Historic Variability for Upland Vegetation in the Medicine Bow National Forest, Wyoming

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1. INTRODUCTION

Because of new information, changing public attitudes, and various congressional mandates pertaining to environmental protection and biodiversity, land managers now face the challenges of ecosystem management (Keiter 1989, Kaufmann et al. 1994, Grumbine 1994, Covington and DeBano 1994, Christensen et al. 1996, Vogt et al. 1997, Kaufmann et al. 1998, Landres et al. 1999). The goals of ecosystem management include maintaining the natural diversity of both species and ecosystems while providing resources desired by humans (Grumbine 1994). Proponents of this approach recognize that our current knowledge of ecosystems is incomplete (Christensen et al. 1996), but they also contend that resource managers should utilize more fully the available scientific information. Also, as more is learned about the effects of certain activities, management strategies can change through a process known as adaptive management (Walters 1986, Christensen 1996). One approach for synthesizing ecological information for an area is through analyses of the historic range of variability (HRV) for key ecosystem variables, such as we have done in this report for the Medicine Bow National Forest (MBNF) in southcentral Wyoming.

An example of how resource management attitudes have changed in the last fifty years is the replacement of the word “stability” with “resilience” (Holling and Meffe 1996). Scientists and managers now recognize that ecosystems have developed with periodic disturbances such as fires and insect epidemics (Botkin 1990, Landres et al. 1999, and others), and that many ecosystems are in a state of recovery, not stability. Disturbances are believed to have perpetuated or even enhanced the biological diversity that is now appreciated by many (Attiwill 1994). In outlining the conceptual framework for ecosystem management for the Forest Service, Kaufmann et al. (1994) concluded that “ . . the most sound basis for ecosystem management is to assure that the variation characterizing [managed] ecosystems includes the range of conditions that are expected at various scales in ecosystems uninfluenced by humans.” The focus is on variability.

Various terms have been used when discussing variability, including “range of natural variability” (Swanson et al. 1993), “natural variation” (Holling and Meffe 1996), “reference conditions” (Kaufmann et al. 1994), “reference variability” (Manley et al. 1995), and “historic range of variability” (Morgan et al. 1994). We adopted “historic range of variability” because it avoids the potential problems associated with interpreting the word “natural.” For this report, we interpret the historic range of variability (HRV) as the spatial and temporal variation in composition, structure and function experienced in an ecosystem from about 1600 to 1850, when the influences of European-Americans were minimal in the MBNF.

A challenge for HRV analyses is to draw conclusions or inferences based on very little data. Several approaches are used. First, biologists and ecologists often are able to make defensible judgments about the conditions that likely existed in the past based on studies of plant adaptations. Knowledge about the organisms present in an ecosystem today provide clues about environmental conditions that existed in the past. Secondly, judgements about the past can be made from different kinds of historical records, such as the journals and photographs of early explorers and the information contained in tree rings. Finally, valuable information is obtained by examining modern reference areas that have not been heavily influenced by European-Americans, such as wilderness areas and some national parks. In fact, no landscape is entirely free from human influences. Native Americans have caused some changes for thousands of years, and even the largest national parks have been affected by technological developments.

HRV analyses require careful attention to four principles. First, variables should be evaluated over at least two spatial scales. The variability that occurs in small tracts of land, commonly referred to as "stands," is quite different from that which occurs over larger areas, referred to as "landscapes." Stands are tracts of forest or other vegetation that are relatively homogeneous in terms of environment, plant species composition, and disturbance history. They generally occupy an area ranging from a few hectares to several hundred hectares. In contrast, landscapes range from several km^2 to several thousand km^2, or more, and they have considerable heterogeneity. The HRV of some variables, such as dead standing trees (snags), is typically higher at the stand scale than at the landscape scale (Fig. 1). Of course, some variables pertain to only one of the two scales. For example, the proportion of a landscape in different cover types is not meaningful at the stand scale.

Pertinent to considerations of scale is the fact that, just as there will be several to numerous stands in a landscape, there are several landscape types in a national forest. For this report, we separated high-elevation landscapes from low-elevation landscapes, with much of our discussion focused on forests and woodlands dominated by lodgepole pine, subalpine fir and Engelmann spruce at high elevations and ponderosa pine, limber pine and Douglas-fir at low elevations. We also present discussions of aspen forests and several types of shrublands, grasslands, and forblands. Forests are emphasized because there is much more historical information available for them than for other vegetation types. Wetlands, including the riparian zone, are mentioned only briefly in our report, as a separate HRV analysis for these important ecosystems will be done in the future.

The second principle is that HRV analyses should consider variability in space as well as time. This principle is difficult to follow because historical data are rarely available for more than one watershed or landscape. However, it is possible to characterize spatial variability in vegetation patterns across management units on the MBNF, as it exists today, and to compare that variability to the range of expected conditions during the HRV period based on our current understanding of ecosystem dynamics. By necessity, HRV analyses must often be deductive in nature—drawing conclusions about specific ecosystem variables based on general knowledge about ecosystems.

The third principle is that “variability” can be characterized in several ways. To illustrate, variability can be thought of as the absolute range of an ecosystem variable or parameter, where extreme and rare conditions are considered (Fig. 2). A second approach involves the calculation of the standard deviation, though we have not found data to calculate this statistic for any variable on the MBNF. Alternatively, the HRV can be based on an estimate of the range of means for a variable over several consecutive periods of time, an approach that ignores extremes (see Fig. 2). HRV analyses based on average conditions tend to produce a narrower range than when based on extremes (see Fig. 1). In this report, we use the range of means approach—conceptually if not quantitatively.

The fourth principle affecting HRV analyses pertains to climate change through time. There is no question that the climate has changed since Rocky Mountain glaciers began to recede about 10,000 years ago. It also is clear that the climate of the 1600s, 1700s, and much of the 1800s was different than at the present time (Millar and Woolfenden 1999). Working in the Bighorn Basin of Wyoming, Gray et al. (in preparation) concluded from tree-ring data that, while the mean annual precipitation was about the same during the 1600s and 1700s as it is today, there are now fewer extended wet and dry periods and the extremes of the wet and dry periods in the 1900s are less. Moreover, preliminary research suggests that the climate of the Bighorn Mountains began to warm in the late 1800s (Christopher Fastie, personal communication). While the HRVs of some ecosystem variables for the reference period 1600 to 1850 may be less applicable to present conditions because the climate has changed, we believe they still provide a useful context for management decisions. Distinguishing the effects of climate change from the effects of European-Americans is a difficult but important challenge for HRV reports.

To characterize the HRV, the historic time period must also be specified. Manley et al. (1995) indicated that, ideally, the period should cover an evolutionary time scale (i.e., as long as possible within the past 10,000 years or so). However, glaciation and the magnitude of climate change in the MBNF during the last 10,000 years created extremely broad ranges of variability of little relevance to the kinds of decisions faced by forest managers today. We selected 1600 to 1850 as the reference period, as it reflects conditions prior to the substantial influences of European-Americans in the area over a time scale relevant to forest management.

Notably, an evolutionary perspective is adopted in our report, but in a different sense than implied by Manley et al. (1995). Some of our interpretations are based on the prevailing consensus about the environmental conditions that must have led to the development of certain adaptations exhibited by the dominant plants of the MBNF. Such adaptations surely influence where species survive today. For example, the thick bark of ponderosa pine (Pinus ponderosa) suggests that the environment where it occurs, or has occurred in the past, is characterized by periodic surface fires. Similarly, the thin bark and serotinous cones of lodgepole pine (Pinus contorta) suggest that the species has been subjected to periodic stand-replacing crown fires.

Estimates of HRV are inevitably limited by our ability to interpret the nature of past ecosystems and determine what was "natural" or "normal" during the reference period (Swanson et al. 1993, Holling and Meffe 1996). Because ecosystems change dramatically over time scales of a thousand years or more, or even a few centuries, we only capture a "snapshot in time" if our reference period is not long enough (Swanson et al. 1993). For such reasons, estimates of HRV are not a panacea. Rather, they serve as one of several tools to aid managers in achieving the societal goal of conserving biological diversity while providing the benefits of various natural resources in a sustainable way.

To be useful to land managers, HRV analyses should attempt to include quantitative ranges for actual variables related to ecosystem structure, composition, or function (Manley et al. 1995). By considering meaningful variables during defined time periods, land managers have an improved context in which to evaluate their decisions. If a management activity appears to push an ecosystem variable to a higher or lower level, based on the kind of evidence included in HRV reports such as this one, then there is reason to evaluate the environmental impacts of that activity—or even consider adopting a different approach. When quantitative data are not available, the goal of specifying variability must be done using qualitative information. For some observers, this quasi-quantitative approach is not satisfactory. However, deductive analyses can provide insights that are not available in any other way and which improve the chance that management decisions will not impede an ecosystem's inherent resilience following the inevitable disturbances that will occur, whether caused by fires, wind, insects, pathogens, or humans.

Landres et al. (1999) concluded that, properly done, HRV analyses provide context and guidance for managing ecological systems. They also emphasized, however, that it is appropriate and unavoidable for HRV reports to include professional judgments, even though based on limited data. Given that fact, they recommended that the authors of such reports should specify their assumptions and value judgments. Briefly, our assumptions and judgments are: 1) there is much yet to be learned about how to manage national forests to meet the various mandates that have led to the emergence of ecosystem management, 2) there is value in the periodic evaluation of traditional management approaches to determine if they are appropriate for meeting an objective, 3) some parts of a national forest are suitable for the careful production of commodities, while others are not—a policy that has been implemented for many years through the designation of, for example, lands suitable and not suitable for timber harvesting—and 4) the information synthesized in this report, in conjunction with site specific information and new studies as they become available, can provide guidance in making land management decisions.

HRV analyses also should not be viewed as a step toward creating "pristine" conditions (Swanson et al. 1993, Allen and Hoekstra 1994). It has been argued, in fact, that ecosystems may be sustainable outside the HRV of a variable if humans are willing to engineer functions once carried out naturally (Allen and Hoekstra 1994). The question remains, however, whether or not humans know enough to create substitutes for natural processes that now are accomplished by thousands of organisms representing many species that occupy ecosystems over large areas—and can afford to pay for them in perpetuity. Notably, national forests already have some features that are unprecedented and clearly outside the HRV, such as roads and dams (Swanson et al. 1993).

In this report for the MBNF, we have attempted to provide 1) an estimate of the HRV for key elements of terrestrial ecosystems, including quantitative ranges for specific variables whenever possible; and 2) a comparison of our HRV analysis with current conditions on the MBNF or conditions that have prevailed since European-Americans arrived. Our focus is on upland vegetation dynamics and plant-related ecosystem variables at the stand and landscape scales; another report is planned for wetland ecosystems. After describing our methods, we review the physical, ecological, and cultural characteristics of the MBNF before proceeding with an HRV analysis for key variables in high-elevation landscapes, low-elevation landscapes, aspen forest, and non-forest vegetation. Our conclusions about apparent human-caused deviations from the HRV are summarized in the last section.

2. METHODS

Estimating the HRV for biological systems should include the following steps (Manley et al. 1995): 1) determine key ecosystem elements (i.e., components, structures, and processes); 2) identify measurable environmental indicators for these elements (e.g., tree density and mean disturbance return interval); and 3) estimate values for the indicators over the selected HRV reference period. We tried to be more inclusive than exclusive in selecting key elements, as the importance of some ecosystem features or processes may not become obvious until the ecosystem is subjected to some kind of stress (Holling and Meffe 1996). HRV studies are a relatively new endeavor and methods are changing rapidly as new information becomes available.

Our approach was to first make plausible judgments about the conditions that must have existed in the past based on current knowledge about ecosystem structure and function. Knowledge about the organisms that exist in an ecosystem today provides clues about environmental conditions that existed in the past, and the general principles of ecosystem structure and function provide a basis for drawing tentative conclusions, deductively, about a specific area. Such conclusions should be considered as hypotheses requiring further testing.

Second, we made judgments about the past from different kinds of historical records, such as contained in tree rings and the journals and photographs of early explorers. Specifically, we used 1) historical records and studies covering approximately the last century that provided quantitative or semi-quantitative information; 2) historical accounts, which provided mostly qualitative information back to the late 1800s; and 3) reconstructive studies from tree-rings, pollen analysis and other methods that provide the only quantitative information for periods prior to 1850 (Kaufmann et al. 1998).

Third, valuable information was obtained from studies on the long-term dynamics of ecosystems in modern reference areas that have not been heavily influenced by humans, such as wilderness areas and some national parks in the region. Yellowstone National Park (YNP) has been especially useful because of the research that has been done there on fire/vegetation interactions. The climate of YNP and the Medicine Bow National Forest are similar, though YNP is generally slightly cooler and more moist (based on data from Martner 1986). The geology of the two areas is different, but the predominant vegetation is similar. Using data from any area as a proxy for another must be done with caution, but to ignore such pertinent information needlessly limits the applicability of research.

Our analyses are based primarily on the compilation and synthesis of information from published and unpublished sources relating to the history and ecology of the MBNF specifically and the Rocky Mountain region in general. Funds were inadequate for initiating new field work. Information was derived from all available sources, including journals, books, USDA Forest Service publications, MBNF reports, and MBNF databases. Much of the spatial and quantitative information for the MBNF is from the Resource Information System (RIS), a database maintained at the Supervisor's Office in Laramie (USDA Forest Service 1989). There are some limitations to the RIS data, but this source provided the most comprehensive spatial information. Of particular concern are the stand-age data, which were determined with small sample sizes.

Several USFS reports were very useful. Specifically, the 1996 specialist report on biophysical and historical aspects of species and ecosystems on the

MBNF, prepared by former MBNF employees Judy von Ahlefeldt and Clay Speas, was a valuable source of information. They summarized data from the RIS database, other internal MBNF documents (e.g., fire records), and the Medicine Bow Collection at the University of Wyoming's American Heritage Center in a way that had not been done previously. Chapters in their report on the geographic and climatic setting, historical ecology, and current terrestrial vegetation were especially helpful.

Another noteworthy document was the 1986 report by Dennis N. Grasso, Scott Thybony, Robert Rosenberg, and Elizabeth Rosenberg. They wrote a "Cultural History Overview" for the MBNF, which later was condensed into a book, “The Medicine Bows” (Thybony et al. 1986). More recently, M. Boglioli, K. Hejtmanek, and N. Stroup (2002) completed a report for the USFS on the history of land use on the Medicine Bow National Forest.

Also, several HRV assessments had been drafted for other national forests in the Rocky Mountain Region when we began our work in 1999 (including the Routt, Rio Grande, San Isabel, and White River National Forests), and later Veblen and Brown (2001) provided an HRV assessment for the Arapaho and Roosevelt National Forests. These reports were helpful as well. Our report, however, is the first attempt for the MBNF.

For this report, we have drawn conclusions about whether existing conditions in terrestrial ecosystems have changed from conditions during the HRV reference period (1600 to 1850) due to the management activities of European-Americans. We defined HRV as the range of means of a variable for several consecutive time periods from 1600 to 1850 (see Fig. 2). In practice, rarely are the data adequate to calculate the HRV in this way. To partially offset this deficiency, "proxy" data were sometimes used, as follows:

1) We reviewed information for the HRV period resulting from studies in other places in the Rocky Mountains.
2) We used the existing pattern (20th century) of "natural" disturbances, such as for blowdowns and stand-replacing fires, as the pattern that must have

existed during the HRV period for comparison to patterns created by human-caused disturbances, such as a timber harvest. Humans surely have modified some of the natural disturbances. Therefore, we only use this method when the modification in the pattern of natural disturbances is thought to be small relative to the modification caused by the human-caused disturbances.

3) We used existing conditions (20th century) in Yellowstone National Park as the conditions on the MBNF during the HRV period. The park's vegetation has been less strongly influenced by European-American activities and may reflect historical patterns that occurred in the Medicine Bow Mountains. In some cases involving fire regimes, we only used Yellowstone data for the period in which fire suppression was not attempted in the park, thereby making the data more relevant to National Forests where attempts to control fires have been the norm.

When data were not available, we substituted information about ecosystem processes and the history of the area to estimate the direction and magnitude of changes from the HRV that seemed probable.

For each variable, we attempted to address 1) its HRV at the stand and landscape scale; 2) its spatial variability across the landscape as well as its temporal variability, when such information was available, and 3) our confidence level (low, medium and high) in concluding whether or not the variable under existing conditions probably is beyond the HRV or trending in that direction. Our conclusions and confidence levels are summarized in Section 8, which includes a summary table that gives the page numbers where each variable is discussed (Table 7).

3. THE MEDICINE BOW NATIONAL FOREST

3.1. Physical setting

The Medicine Bow National Forest covers approximately 405,000 hectares (1,000,000 acres) and is composed of four geographically separate units on the north end of the Southern Rocky Mountain physiographic province (Fig. 3). The two eastern units are in the Laramie Mountains—the small 22,267 ha (55,000 acre) Sherman Mountains unit between Laramie and Cheyenne (part of the Laramie District; sometimes called the Pole Mountain unit), and the moderately sized 72,874 ha (180,000 acre) Laramie Peak unit toward the north end of the Laramie Mountains (the Douglas District). To the west of Laramie and east of Saratoga is the Snowy Range unit of the Medicine Bow Mountains (part of the Brush Creek/Hayden District and most of the Laramie District), which is the largest unit and covers about 210,526 ha (520,000 acres). To the west of Encampment and east of Baggs is the Sierra Madre unit (part of the Brush Creek/Hayden District), a northern extension of the Park Range in Colorado that covers approximately 137,651 ha (340,000 acres).

All of the MBNF is mountainous, with elevations ranging from 1915 m (6,300 ft) near Laramie Peak to 3652 m (12,013 ft) on the top of Medicine Bow Peak in the Snowy Range (Fig. 4). The Snowy Range is the most massive unit and is bordered on the east by the Laramie Basin and on the west by the North Platte River Valley. Both the Snowy Range and Sierra Madre have cores of Precambrian granite and gneiss, but these rock types are overlain in many areas by metamorphosed sedimentary rocks, quartzite, and schist (Houston 1968, Houston and Ebbett 1977, Price 1973, Oviatt 1977). A mountain glacier formed in the vicinity of Medicine Bow Peak during the Pleistocene, which resulted in the formation of cirques, till fields, moraines, and numerous lakes. Glaciation in the Sierra Madre was much less extensive, though Battle Lake occurs at the base of a cirque and Bridger Peak is thought to have been a monadnock above the ice. No glaciers formed over the Laramie Mountains.

The Laramie and Sherman Mountains units comprise the northernmost extension of the Colorado Front Range. The topographic relief is less than the Snowy Range and Sierra Madre units, but they all have similar geologic origins. Metamorphosed sedimentary and volcanic rocks are common. Mafic intrusive rocks are found in the area, and the Precambrian Sherman granite is abundant, such as in the Vedauwoo recreational area. Sandstones and limestones of the Casper and Fountain formations are found on the west slopes in the Sherman Mountains. To the east are Oligocene sedimentary substrates, remnants of which form the “gangplank” that slopes toward Cheyenne. An outcrop of limestone occurs along parts of Horseshoe Creek.

As with other portions of the Rocky Mountains, the climate of the MBNF varies greatly with elevation and topographic position. Generally, the mean annual precipitation ranges from 38 cm (15 inches) at lower elevations to 100 cm (39 inches) or more in the subalpine zone (Martner 1986). Over much of the forested area above 2600 m (8580 ft), about two-thirds of the precipitation falls as snow. The Laramie Mountains unit is lowest in elevation and the warmest, with a mean annual temperature of 7° C (45° F) and a maximum recorded range of -40° C (-40° F) to 43° C (110° F) (Martner 1986). In the subalpine and alpine zones, the mean annual temperature is less than 2° C (35° F). Frost can occur at high elevations during any month of the year, with a low of -46° C (-50° F) in the winter and a maximum of about 32° C (90° F) on very warm days at mid-elevations in the summer (Martner 1986, Marston and Clarendon 1988). South slopes are considerably warmer than north slopes or creek bottoms. The numerous watersheds function also as cold-air sheds, creating occasional temperature inversions and contributing to great spatial variability in the elevational distribution of different vegetation types. The Laramie and Sherman Mountains are lower and drier than the Sierra Madre and Snowy Range units, and the southern half of the Snowy Range unit is drier than the northern half.

Historically, the Rocky Mountain region has experienced great climatic and geological changes over time scales of millions of years (Table 1). Consequently, the vegetation has changed also. By the beginning of the

Quaternary period, approximately 2 million years ago, many of the species present today were in place (Tidwell 1975). From 2 million to 10,000 years ago, a period known as the Pleistocene epoch, several glacial advances and retreats affected both the topography and vegetation in the northern part of the Snowy Range and portions of the Sierra Madre (Marston and Clarendon 1988). Since the end of the Pleistocene around 10,000 years ago, shifts in climate have continued, often triggering changes in the spatial and elevational distribution of plant species (Table 1; Whitlock 1993). During the last 500 years, the climate of the region has been characterized by extended, severe droughts as well as by extended wet periods (Gray et al., in preparation). Over shorter time scales (2-5 years), the El Nino Southern Oscillation (ENSO) has influenced climate, and subsequently fire occurrence, in the Rocky Mountains. In Colorado, El Nino years appear to have been wetter than average and La Nina years have been drier (Veblen et al. 2000). A similar pattern probably extends into the MBNF of southern Wyoming, though it weakens from south to north.

3.2. Ecological setting

The dramatic changes in climate over time have important implications for HRV analyses. Foremost, we must recognize that, by considering variation observed during our reference period (1600-1850), we are considering only a short period in the ecological history of MBNF ecosystems. Soils, for example, surely have developed over thousands of years, not centuries. Also, the mean return interval for various kinds of disturbances during our reference period, including fire, undoubtedly have changed as the climate changed.

Topographic and climatic diversity, along with periodic natural disturbances, caused a diversity of vegetation types to occur across the MBNF during the reference period. The zonation of vegetation on the mountain ranges is similar to other parts of the Rocky Mountains, but there is at least one important difference: The low-elevation woodlands of ponderosa pine and Douglas-fir common in Colorado and other parts of the Rocky Mountains are limited to small areas in the MBNF, probably because of the comparatively high elevations of the surrounding plains. To illustrate, foothill grasslands and shrublands typically grade directly into montane forests dominated by lodgepole pine around the Snowy Range and Sierra Madre units (Knight 1994a). Ponderosa pine is common only at lower elevations in the Sherman and Laramie Mountains. Another notable feature is the extensive aspen forest on the southwest slopes of the Sierra Madre Mountains, from the foothills up to about 2,590 m (8,500 ft) elevation. Extensive aspen forests are common to the south in Colorado (Peet 1988), but they occur in much smaller groves over most of the MBNF and Wyoming.

Various classification systems have been applied to the upland vegetation (Table 2). At the broadest scale, almost all of the MBNF is in the Northern Parks and Ranges Section (M331I) of the Southern Rocky Mountain Steppe-Open Woodland-Coniferous Forest Province (Bailey et al. 1994, McNab and Avers 1994). The one exception is the western portion of the Sierra Madre, which is part of the Central Basin and Hills Section (342F) of the Intermountain Semi-desert Province. At an intermediate scale, the forest vegetation can be classified by the dominant overstory species, i.e., cover type, which is the approach adopted for the MBNF RIS database (Fig. 5). The primary cover types are lodgepole pine, Engelmann spruce-subalpine fir, ponderosa pine, aspen, limber pine, and Douglas-fir forests, plus upland shrublands (e.g., sagebrush, serviceberrry, and mountain mahogany shrublands) and grasslands or meadows (dominated by various species of grasses and forbs).

Most of the forested land occurs in the Snowy Range and Sierra Madre units, where about 78% of the land is forested and about 65% of the forests are dominated by lodgepole pine, Engelmann spruce and subalpine fir (Table 3 and Figs. 5a and 5b). On the Sierra Madre unit, aspen occupies about 17% of the land area, much more than in the Snowy Range, Laramie Peak, and Pole Mountain units (3.8, 1.4, and 7.4%, respectively). Douglas-fir occupies no more than 2% of the land area in any of the four units (Table 3 and Fig. 5), but ponderosa pine is most abundant on the Laramie Mountains unit (47% of the land area; Fig. 5d). Limber pine is most common in the woodlands around

Laramie Peak and Pole Mountain, but occupies no more than about 5% of the MBNF.

Overall, approximately 55% of the forested land on the MBNF was classified as suitable for timber harvesting in the 1985 management plan. On the Sierra Madre and Snowy Range units, respectively, approximately 15% and 30% of the forests are more than 150 years old (Fig. 6), suggesting that about 85% and 70%, respectively, of the forests in these units have been subjected to burning or timber harvesting since 1850.

Shrublands and grasslands are common throughout the MBNF (Table 3 and Fig. 5). Big sagebrush dominates most of the shrublands, but on upland sites there are small areas dominated by mountain-mahogany and juniper. Willows and thinleaf alder occur along the numerous creeks. The northern extension of Gambel oak occurs in the southwestern foothills of the Sierra Madre unit, where it is common in a small area (about 620 ha or 1530 acres). Grasslands are the most common vegetation type on the Sherman Mountains unit (Table 3 and Fig. 5c), where they occupy 47% of the landscape. In the other three units, grasslands and meadows cover no more than 3 to 8% of the land area. The most extensive alpine zone is in the Snowy Range unit (3,483 ha or 8,601 acres), but it comprises only 1.6% of the land area even there. Surface water (lakes, reservoirs and streams) occupies less than 0.5% of the landscape in all units (Table 3).

More specific forest classifications for the MBNF are based on both overstory and understory species. Alexander et al. (1986) described nine principle forest series, based on overstory dominants, and 16 habitat types, based on overstory dominants and understory indicator species (after Daubenmire and Daubenmire 1968; see Table 2). Proponents of the habitat type approach contend that, while the spatial distribution of each tree species is controlled by factors such as elevation and topographic position, more specific environmental conditions (i.e., habitat types) are better reflected by understory species. Considering both forest and non-forest vegetation, approximately 307 habitat types and community types have been identified on the MBNF (von

Ahlefeldt and Speas 1996). Unfortunately, they have not been mapped and little information is available for each one.

After considering the appropriate scale to address HRV, we organized our discussion around four landscape types—subalpine or upper montane, high-elevation forests; montane forests at low elevations; aspen forest; and non-forest vegetation (namely, upland shrublands and grasslands/meadows). There are important ecological distinctions between different habitat types and community types within each of the four landscape types that we examined, but the differences blur over the long time periods of HRV analyses. Furthermore, because of successional relationships among the most common overstory species (lodgepole pine, subalpine fir and Engelmann spruce), it is difficult to discuss HRV in isolation for each forest type.

3.3. Cultural setting

For centuries, humans have lived and traveled throughout the area that is today the MBNF (Grasso et al. 1981, Fawcett and Francis 1981, Thybony et al. 1986, Boglioli et al. 2002). Consequently, managers have inherited the legacies of past land uses and are not starting with a “blank slate” (Romme et al., 2000). This is an important consideration for several reasons. First, our ability to interpret pre-European-American conditions on the MBNF will be limited by events that have happened during the past 150 years. Also, it is difficult to distinguish events that were “natural” from those caused by Native Americans. Indeed, it may be necessary and even desirable to include some human influences as a factor affecting the HRV. However, many of the unprecedented human influences of European-Americans present unique challenges for managers now attempting to bring conditions within the HRV.

For centuries, Native American tribes such as the Northern Arapahoe, Northern Cheyenne, Ogalala Sioux, Ute, and Eastern Shoshone either inhabited lowlands surrounding the forest or used canyons and mountain passes as travel routes (Grasso et al. 1981, Thybony et al. 1986). Boglioli et al. (2002) summarize information about the Native American tribes that occupied the area for centuries. Their impacts on the landscape are largely unknown but probably were minor. Only small amounts of wood, usually from small trees, would have been harvested. Fires would have been started in the vicinity of travel routes and campsites in the low-elevation forests and woodlands (Barrett and Arno 1982, Gruell 1983a), and some of those fires might have burned into adjacent high-elevation mountain vegetation as well. However, Baker (2002) concluded that, for Rocky Mountain landscapes in general, Native Americans had little effect on fire regimes in lodgepole pine and spruce-fir forests.

Beginning in the early 1800s, trappers and explorers from the east began to move into the Rocky Mountain West. Human land uses on the MBNF during this time probably remained about the same, primarily because travelers in southern Wyoming usually followed courses around rather than through the mountains (Townsend 1978, Grasso et al. 1981). Perhaps the most significant additional influence was the cascade of changes resulting from the reduction in beaver populations caused by trappers, plus a large fire in the southwestern Sierra Madre Mountains (Grasso et al. 1981, Thybony et al. 1986). In general, the effects of Native Americans and the early explorers were considerably less severe than those of European-American settlers after the 1860s, when the harvesting of big trees became common.

By the mid-1800s, the Overland and Oregon Trails facilitated the immigration of European-Americans to southern Wyoming. In the decades that followed, millions of large trees were cut for telegraph poles, railroad ties, mine timbers, fuel wood, and construction (Figs. 7, 8, and 9). Lodgepole pine trees with a diameter at breast height (dbh) of 28-36 cm (11-14 inches) were harvested for railroad ties, a practice known as “tie hacking” that continued until the 1950s (Grasso et al. 1981, Thybony et al. 1986, Wroten 1956). Human-caused fires burned through the slash of some of the harvested stands, often killing the trees that were not harvested. The slash often was used as a source of fuel wood for cooking and heating. Riparian zones fringing major creeks, like Douglas Creek, were sometimes clearcut to facilitate the movement of logs and railroad ties to rivers, where they were floated downstream (Young et al. 1989, 1994). In the

Sherman Mountains there were insufficient numbers of trees to provide an adequate supply of ties, so they were imported from outside the region (Wroten 1956). Many trees in the Pole Mountain area were cut, though much of the land was not forested during the reference period (1600-1850). To the north, near Laramie Peak, trees were cut for telegraph poles and other purposes (Thybony et al. 1986).

Early European-Americans also took advantage of other resources. Mining for gold and other ores became common in portions of the Snowy Range, Sierra Madre and Laramie Mountains (Duncan 1990). Old photos show large areas of clearcut timber prior to 1880 in the vicinity of mines, especially in the Sierra Madre unit (Thybony et al. 1986). Also, large herds of sheep and cattle were common by 1880 on mountain grasslands and shrublands (Grasso et al. 1981). The livestock sometimes benefited from the increased forage production that occurs following forest fire and timber harvesting. Simultaneously, the native herbivores (elk, deer, bighorn sheep, pronghorn antelope, and bison) were hunted for eastern markets and sport (Dary 1986), driving some species to near extinction. Poisoning and shooting decimated populations of the native predators, especially wolves, mountain lion, and grizzly bear. Many parts of the MBNF are thought to have burned during the late 1800s, with ignitions commonly associated with wood stoves, steam-powered saw mills, and the Laramie Hahn’s Peak and Pacific Railroad that crossed the Snowy Range. Partially due to harvesting and burning in the late 1800s and early 1900s, approximately 70-80% of the forests on the MBNF are now 90 to 130 years old (MBNF RIS).

Concerns about the adverse effects of European-American activities brought about the period of National Forest administration. In 1902, the U.S. government began the establishment of several forest reserves in southcentral Wyoming that would eventually become the MBNF (Medicine Bow National Forest 1905). With the establishment of the reserves, land uses such as timber harvesting and grazing were regulated and more careful attention was given to finding the best ways to use the land (Pinchot 1907, Armstrong 1935). Also, fire suppression became a primary concern. Fire was personified as the “enemy” to forest managers (USDI 1897, p. 80). Steam-powered sawmills prior to 1903, and gasoline-powered mills later, were a common source of ignitions.

In the mid-1930s, harvests of 65-85% of the trees in a stand became common. Partially because of problems with blowdown in thinned stands, clearcutting was initiated in the late 1940s (Lexen 1949, Alexander 1975, Alexander and Edminster 1980). Also, new levels of prosperity in the region after World War II created larger markets for timber than ever before and the creation of new stands of rapidly growing, economically valuable trees became a common goal for National Forests (Langston 1995)—providing another incentive for clearcutting. The cutting of larger trees was encouraged. Gasoline-powered sawmills, some that were portable, had been in use since the 1920s and were common. Also, road building increased after 1950, making much more of the forest accessible for timber harvest (Fig. 10). In addition to strip cutting and dispersed clearcuts, some stands were partially cut by individual selection, group selection, and shelterwood silvicultural systems (Alexander 1975, Alexander and Edminster 1980). Unlike the previous harvesting that was concentrated in low-elevation stands, many subalpine forests were cut for the first time in the 1950s and 60s. Natural tree regeneration in harvested stands has been quite good on the MBNF, except on wind-swept sites at high elevations. In such environments there may be less seed, and seedling mortality during the winter can be high (probably because of winter desiccation that occurs when too much of the snow is blown into the forests downwind). Even there new trees eventually become established, though their density is low and their growth is slow. Over most of the MBNF, seeding or planting of trees is not required in harvested areas to assure an adequate number of seedlings within the 5-year period mandated by the National Forest Management Act of 1976 (Jerod Delay, personal communication)

Timber sales in the MBNF have declined in recent decades, from a high of ~58 million board feet per year in the late 1950s to ~6 million board feet in 1999 (MBNF RIS). Clearcut or patch-cut harvesting is still common, though with greater sensitivity to wildlife and aesthetic values. The intensity of livestock grazing has been reduced as well, from about 400,000 sheep and 200,000 cattle in the early 1900s (Boglioli et al. 2002) to about 20,000 sheep and 16,000 cattle at the present time (MBNF RIS).

An important new effect has been the influence of people from throughout the nation who are engaged in a wide range of recreational activities. Moreover, new homes have been built on acreages adjacent to the MBNF. Due to accessibility, the popularity of outdoor recreation, and proximity to large and rapidly growing human populations in Colorado, such influences can have widespread impacts. Managing outdoor recreation and exurban developments could become as controversial as timber management. As noted, the increasing demands placed on public lands and new mandates from Congress during the last 30 years have led to the philosophy of ecosystem management, for which HRV assessments are desired.

4. THE HRV OF HIGH-ELEVATION FORESTS

4.1. Variation in species composition and succession

Forests dominated by lodgepole pine, Engelmann spruce and subalpine fir are common at middle to high elevations on the Snowy Range and Sierra Madre Mountains (Figs. 11 and 12) and have been the subject of numerous studies. They are commonly known as upper montane or subalpine forests. Lodgepole pine is the dominant species at elevations of approximately 2,400 to 3,000 m (7920 to 9900 ft), where it occupies 66% and 51% of the forested area on the Snowy Range and Sierra Madre, respectively (von Ahlefeldt and Speas 1996). Subalpine fir and Engelmann spruce are dominant in the subalpine zone (3,000 to 3,300 m; 9900 to 10,890 ft), covering 27% of forested land on both the Snowy Range and Sierra Madre (von Ahlefeldt and Speas 1996). Notably, subalpine fir and Engelmann spruce occur throughout the elevational range of lodgepole pine and all three species often occur together (Veblen 1986). In the Laramie Mountains, lodgepole pine is less common (31% and 37% of the forested land on the Sherman Mountains and Laramie Mountains units, respectively), and Engelmann spruce and subalpine fir occur only in small stands.

Aside from elevation, the distribution of tree species is affected by moisture availability, which usually is controlled by elevation and topographic position. Spruce-fir forest is typical on mesic sites, such as in ravine bottoms or on north slopes, while lodgepole pine is common on comparatively dry sites, such as on south slopes (Romme and Knight 1981). Small patches of spruce-fir forests can be found in ravines at lower elevations where cold air drainage occurs (Romme and Knight 1981). High-elevation landscapes often have abrupt changes in topography, causing many forests to occur in small patches (Knight 1994b, Knight and Reiners 2000). Notably, the topography of some parts of the MBNF is comparatively flat, enabling the development of rather extensive forest stands that are accessible for timber harvesting and susceptible to fragmentation, especially in the Snowy Range (Marston and Clarendon 1988, Knight and Reiners 2000).

Lodgepole pine traditionally has been considered a pioneer species because of its relative intolerance for the forest understory environment and because it grows well following disturbances (Baker 1949, Lotan and Perry 1983, Lotan et al. 1985, Parker and Parker 1994). Its seedlings are able to withstand cold temperatures like those found on most of the Snowy Range, the Sierra Madre Mountains, and at higher elevations on Laramie Peak (Cochran and Berntsen 1973). This characteristic, combined with a history punctuated by disturbances over the last 150 years, help explain the prevalence of lodgepole pine on the MBNF today. Lodgepole pine is a disturbance-adapted species, often bearing cones that remain closed for many years, thereby storing large numbers of seed on a tree. Known as serotinous or “closed” cones, they open primarily when exposed to higher than normal temperatures, such as during a fire or when the cones are near the soil surface (which is warmer than the canopy). Notably, not all lodgepole pine produce serotinous cones and the proportion of trees with "closed" and "open" cones is highly variable (Lotan and Perry 1983, Muir and Lotan 1985, Tinker et al. 1994).

As mature lodgepole pine begin to die after a hundred years or more, they sometimes are replaced by subalpine fir and Engelmann spruce—a textbook example of succession that leads to the development of an uneven-aged forest structure (Daubenmire 1943, Oosting and Reed 1952, Knowles and Grant 1983, Knight 1994a). Commonly, another disturbance occurs before spruce and fir become dominant. Notably, this successional pathway is just one of several possible trajectories following stand replacing disturbances (Fig. 13 and Table 4). Relatively stable lodgepole pine forests have been reported on the MBNF (Alexander et al. 1986) and throughout the Rocky Mountains (Moir 1969; Loope and Gruell 1973; Whipple and Dix 1979; Despain 1983, 1990; Lotan and Perry 1983; Peet 1988). Such forests are typically found where conditions are not favorable for subalpine fir or Engelmann spruce seedling establishment (e.g., on sites that are relatively dry) or where there is no readily available seed for these and other species. Many lodgepole pine trees commonly die before reaching 250 years of age, with reproduction occurring in canopy gaps formed by dying trees. Aspen can be another important pioneer species, but, like lodgepole pine, it too forms self-perpetuating stands in some habitats (Mueggler 1985). Elsewhere, spruce and fir may be pioneer species, as Doyle (1997) documented in Grand Teton National Park. Reviewing the full range of successional trajectories is beyond the scope of this report, but a consideration of succession provides a basis for evaluating potential deviations from the HRV of some stand and landscape variables.

Generally, the time required for a spruce-fir forest to develop after a fire may range from 150 to over 1,000 years (Oosting and Reed 1952, Billings 1969, Mueggler 1976, Romme and Knight 1981). Villalba et al. (1994) concluded that "old growth" subalpine forests in Colorado should have some trees greater than 300 years old. Mehl (1992) suggests 200 years. Two or three centuries may be required for the development of important components of old-growth forest, namely, relatively large trees, numerous snags, and relatively high amounts of coarse woody debris (CWD) (Mehl 1992, Rebertus et al. 1992). Franklin and Laven (1989) stress that forest development is not easily predicted by simple models, as disturbances of different timing and severity create alternative pathways of succession, depending on environmental conditions at the time. Indeed, the time between subsequent disturbance events in some areas may be shorter than the time required for old spruce-fir forests to develop. On some watersheds, old spruce-fir forests can be restricted to places protected from fire, such as ravines or leeward slopes (Romme and Knight 1981, Despain and Romme 1991). Notably, Baker and Kipfmueller (2000) could not find a relationship between old forests and topographic position in the 3,241 ha (8,005 acre) Rock Creek watershed they studied on the north end of the Snowy Range unit.

It is important to recognize that secondary succession initiated today could have quite different trajectories than 200-300 years ago because of different climatic conditions. Similarly, the distribution of individual species may have shifted for the same reason. Investigators have found evidence in the pollen record, for example, that the relative abundance of lodgepole pine, Engelmann spruce and subalpine fir at both the stand and landscape scales has shifted over long time periods due to climate change (Hansen 1940, Fall 1997). Whitlock (1993) documented altitudinal shifts of tree species distributions in northwest Wyoming during the last 12,000 years; and in recent centuries lodgepole pine, subalpine fir and Engelmann spruce have grown into meadows (Patten 1969, Schimpf et al. 1980, Pearson et al. 1987, Jakubos and Romme 1993, Hessl and Baker 1997). Some of these observations have been attributed, at least in part, to warmer and possibly somewhat wetter conditions since the end of the Little Ice Age (ca 1850-1870).

4.2. Disturbances in high-elevation forests

A variety of disturbances occur in high-elevation forests of the Rocky Mountains. Historically, fire has been the predominant stand-replacing disturbance, with occasional insect epidemics and windstorms. Apparently, diseases cause less widespread disturbances. As discussed below, large numbers of trees can become infected with dwarf mistletoe infections after long periods without fire, but the effect of mistletoe is usually to slow growth rather than to kill the tree. Successional processes have been initiated over and over during the last 10,000 years, since the retreat of the last glaciers, creating a diverse landscape mosaic. Disturbances have not acted in isolation, but rather they have had interactive relationships that enhance the natural diversity of subalpine forests. For example, a windstorm that topples a large number of trees may increase the probability of an insect epidemic or fire in some areas (Schmid and Frye 1977, Lindemann and Baker 2001).

Today, the frequency, size and intensity of historic disturbances, and the forests that develop after different kinds of disturbances, often have been altered by management activities. In the following sections we examine these changes in more detail and attempt to characterize HRV for the major disturbances. Subsequently, we discuss how some stand and landscape variables appear to be either beyond their HRV at the present time or are trending in that direction.

4.2.1. Fire

Fire has shaped the vegetation mosaic for thousands of years by causing disturbances on a variety of scales. The evidence of historical fires is recorded by charcoal and pollen deposited in bog and lake sediments (Mehringer et al. 1977, Millspaugh and Whitlock 1995, Whitlock and Millspaugh 1996, Fall 1997), and by annual rings on trees that survive fire events (fire scars) or which become established within a few years after the fire. Fire intensity in high-elevation forests varies according to weather, fuel, and site conditions.

A common approach for estimating site-specific fire return intervals (i.e., for stands) involves age-class sampling and evaluation of stand structure and fuel accumulation (e.g., Billings 1969; Romme 1980a, 1982; Romme and Knight 1981; Romme and Despain 1989; Kipfmueller and Baker 1998a, 2000). Based on this approach, the fire-free interval for a particular stand of subalpine forest in the MBNF can be 25 to 700 years. Average fire return intervals have also been estimated by counting the rings on fire-scarred trees (Hawkes 1980, Kipfmueller 1997). Such estimates are often shorter than others because they consider the occurrence of all fire events, not just stand-replacing fires. The great variation in fire-return-interval estimates is most likely related to differences in site characteristics and species composition. The longest fire-free intervals probably are associated with mesic sites found at high elevations, in ravines, or on north slopes (Billings 1969, Hawkes 1980, Romme and Knight 1981, Tinker et al. 1994)—though this pattern is not universal (Baker and Kipfmueller 2000).

Another way of characterizing fire frequency is the mean fire interval (MFI) for a large area, such as a landscape (e.g., Veblen et al. 1994, Wadleigh and Jenkins 1996, Kipfmueller 1997, Kipfmueller and Baker 1998a). MFI is calculated simply as the arithmetic mean of intervals between all known fire events within a landscape, even though different portions of the area burn each time (Romme 1980b). Because this statistic is area dependent, larger study areas typically have more fire events, leading to lower MFI calculations (Arno and Peterson 1983, Kipfmueller 1997). Therefore, MFI estimates from different areas are difficult to compare.

Kipfmueller and Baker (2000) calculated MFI for the Rock Creek watershed (3241 ha) on the north end of the Snowy Range unit for three distinct time periods: 1) prior to the arrival of European-Americans in the 1800s (i.e., the reference period); 2) after their arrival but before effective fire control; and 3) in the mid- to late-1900s when fire control was attempted in some areas. Their results suggest a shortening of the MFI with the arrival of European-Americans, which indicates an increase in ignitions (Fig. 14). The short MFI for the late 1800s may reflect a combination of European-American influences and a drier climate. Since the early-1900s, MFI has increased slightly, but it is still shorter and less variable than before the arrival of European-Americans, a trend noted by Zimmerman and Laven (1984) as well. Therefore, while fire ignition rates were highest during the late 1800s, they probably have remained higher in the 1900s than they were during the reference period as a whole, in part due to the influence of human-caused ignitions. Interestingly, the number of total ignitions (including human-caused) have been much higher in the Snowy Range and Sierra Madre in recent years than in Yellowstone National Park, while the number of lightning-caused ignitions on the two MBNF units has been similar to total ignitions in YNP (Fig. 15). During the same time, however, the amount of land area burned on the MBNF was relatively low compared to YNP (Fig. 16), suggesting that fire suppression on the MBNF has been effective.

Further insight into the frequency and size of fire events at high elevations, namely on the Snowy Range and Sierra Madre units, can be gained by considering both long-term and recent fire histories in Yellowstone National Park. As an area that has remained relatively undisturbed by human influences, and one that is in many ways composed of similar topography and vegetation as the MBNF, YNP provides a useful proxy for HRV analyses. Despain and Romme (1991) reconstructed the fire history for a large, 129,600 ha (320,112 acre) study area in YNP (approximately two-thirds the area of the Snowy Range unit). Although fire apparently has occurred in every decade, they found that large-scale, stand-replacing fires have occurred during four main periods since 1690 AD, separated by intervals of 20, 110, and 118 years. Notably, these data do not represent intervals between fires in a specific stand, but rather between major fire events somewhere in their study area. During each major fire period, a large percentage (7-26%) of the entire area burned, initiating extensive, relatively even-aged patches of forest. Despain and Romme concluded that such even-aged cohorts become increasingly susceptible to large-scale burning as succession proceeds. Very likely they will burn within 200-400 years, whenever weather conditions permit (Table 5). Ignitions are likely to occur every summer, but most fires do not continue to burn.

Recent fire observations in YNP provide information on the variability in size for individual fires in the absence of fire suppression. Since 1972, YNP managers have allowed lightning-caused fires to burn in many instances, providing a proxy for possible dynamics in the early and middle 1800s. Based on fires that burned from 1972-1988, Despain and Romme (1991) observed that lightning fires typically burn only a few hectares or less under most conditions, and that weather conditions in only five years during a 15 year period (1972-1987) allowed fires to burn more than 100 ha (247 acres), namely, once every three years, on average. Looking at the area burned per year per million ha from 1972-1987 in YNP (see Fig. 16), three important trends emerge: 1) The area burned in a single year is highly variable, ranging from 0 to 10,423 ha per million ha; 2) the mean area burned per year per million ha is toward the low end of the range at 1,138 ha/year (2,811 acres/year); and 3) the median, possibly a better indicator of a typical year, is only 12 ha/year/million ha.

Thus, the overall fire regime in YNP is characterized by numerous fires (lightning and human-caused) that are unable to spread over more than a few hectares, and relatively few fires that burn large areas (and have by far the greatest effect on the landscape mosaic, such as the fires of 1988)—a finding consistent with the results of Kipfmueller and Baker (2000) for a small watershed on the north end of the MBNF. The driving factors that create this regime appear to be weather conditions and the type and distribution of fuel (Despain and Romme 1991, Baker, in press). Kipfmueller and Baker (2000) identified a moderately sized fire around 1680 AD that burned at least 2,500 ha (6,175 acres) in the northern Snowy Range (as well as another in 1743 that apparently burned >1,500 ha). Much larger fires probably occurred, as they have in YNP. However, thus far no evidence for them has been found on the MBNF (William Baker, personal communication)—perhaps because a substantial portion of the evidence has been lost due to the extensive timber harvesting and fires that occurred after the arrival of European-Americans.

Fire records for the MBNF from 1945-1993 (summarized by von Ahlefeldt and Speas 1996) suggest that fires have played a much smaller role in the mid- to late-1900s than previously, as only six fires larger than 125 ha (309 acres) burned in the Snowy Range and Sierra Madre during this 48 year time span. Notably, these six fires accounted for almost 70% of the total land area burned (3,831 ha / 5,642 ha) (Fig. 17). The largest fire recorded from 1945 to 1993 in these two areas was only 513 ha, much smaller than large historical fires (e.g., Despain and Romme 1991; Kipfmueller and Baker 2000), and the vast majority of fires that burned during this time (ca 95%) were less than 4 ha (9.9 acres). The total area burned during the 48 year period (1945-93) on the Snowy Range and Sierra Madre was only 5,462 ha, approximately 1.6% of the National Forest land in these two areas. Most of the fires burned at middle elevations (2400-2850 m), where lodgepole pine is the dominant species.

The probable cause for this low incidence of fire in the MBNF is effective fire suppression (Kipfmueller and Baker 2000). Notably, 1988 was a drought year in the MBNF as it was in YNP. Forty-three fires were ignited on the MBNF in 1988, 29 from lightning and 14 from human activity, but all were extinguished before they had burned more than a few hectares, except for one fire that burned 400 ha (988 acres).

Unfortunately, there is insufficient information available to conclude that lodgepole pine forests at lower elevations have a shorter MFI than at higher elevations, though there is a tendency for younger stands to be located at lower elevations in both YNP and the MBNF (William Romme, personal communication). Fires in low-elevation forests, once started, could have spread into the adjacent forests during our reference period, thereby shortening the MFI in forests at higher elevations. However, Baker and Kipfmueller (2001) did not find evidence for a relationship between MFI and elevation in the Rock Creek drainage on the north end of the Snowy Range. Research over a larger area in the MBNF is needed.

Aside from fire suppression, weather probably has been an important factor accounting for the relatively low extent of burning in the MBNF in recent years. Recent research suggests that weather conditions rather than fuel accumulations (Bessie and Johnson 1995, Weir et al. 1995, Clark 1990, Keane et al. 2002) cause large stand-replacing fires, though some stress the combination of weather and fuels. For example, Romme and Despain (1989) concluded, based on fuel availability, that extensive fires could have occurred in YNP after about 1930, but they did not occur until 1988 because of weather conditions. Baker (2002) provides a detailed review of various factors affecting the frequency and extent of fires in the Rocky Mountain region, concluding that climate, fuels and elevation are linked causes and that ongoing human land uses can be confounded with climatic effects.

Dealing with infrequent but large fires presents a difficult challenge to forest managers. On one hand, disturbances of the magnitude of the 1988 Yellowstone fires could be characteristic of our reference period for the MBNF and could have had very important ecological effects (e.g., Christensen et al. 1989). Preventing such events in the future, if that is possible, could have consequences for biological diversity and other ecosystem features (Keane et al. 2002). On the other hand, such disturbances also are a frightening extreme in the HRV that the public has encouraged managers to suppress (Swanson et al. 1993, Manley et al. 1995). Whether or not attempts to suppress intense fires in high-elevation forests have been or can be successful remains controversial. Many fires, whether started by lightning or humans, certainly have been put out, but the suppressed fires might not have affected much land area without the wind and drought that accompanies the often uncontrollable large fires.

Notably, there is little basis for arguing that fire suppression in YNP from 1872 to 1972 had a major effect on the magnitude of the 1988 fires over most of the high plateaus of YNP. While evidence for comparably large fires does not yet exist for the MBNF, it is probable that such fires have occurred in the Snowy Range and Sierra Madre units and that they will occur again, regardless of attempts to control them. In unmanaged forests (such as YNP), stand replacing fires may occur in the same stand, on average, every 200-400 years; and they may burn over a large proportion (perhaps 5-25%) of the landscape approximately once each century. Where landscape patterns and stand structures are modified by humans, as in the MBNF, the intervals are more difficult to predict. Harvesting often is done now with the objective of reducing fuel loads and flammability as well as providing a source of wood. However, wild fires that burn large areas probably are determined as much by drought and wind as by fuel loadings, if not more so (Johnson and Larsen 1991, Bessie and Johnson 1995, Weir et al. 1995, Clark 1990, Baker, in press). Thus, it appears that managed forests at high elevations are just as likely to have large, stand-replacing fires as unmanaged forests.

4.2.2. Insects

Mortality caused by insects has probably been the second most important form of disturbance in high-elevation forests, as the spruce beetle (Dendroctonus rufipennis) and mountain pine beetle (Dendroctonus ponderosae) are capable of reaching epidemic population sizes in some spruce-fir and lodgepole pine forests. As with fire, such epidemics can be detected in the annual growth-ring patterns of trees, providing insights into forest dynamics in the mid-1800s. No published studies of this nature have been done on the MBNF, and therefore our current understanding of the HRV for insect populations must be derived from general observations and studies done elsewhere (Klein et al. 1979, Christensen et al. 1987, Veblen et al. 1994, Veblen 2000). Epidemic severity appears to be highest in the northern Rockies, where millions of trees have been killed in a single year (Klein et al. 1979).

Based on historical reports, the mountain pine beetle (MPB) and other associated bark beetles (e.g., Ips spp.) have not been a significant factor in the lodgepole pine forests of the MBNF during the last century. Recent investigations have found that cold environments, such as at high elevations in much of the Snowy Range and Sierra Madre, may inhibit MPB epidemics (Amman 1977, Amman et al. 1977; Amman 1978, 1989). Only small areas of endemic population levels exist in the higher elevation forests of the MBNF, and no major epidemics have been recorded (von Ahlefeldt and Speas 1996). A MPB epidemic did affect a large number of ponderosa pine at low-elevations in the Laramie Peak area, from 1988 to 1994, but few lodgepole pine apparently were affected at high elevations (von Ahlefeldt and Speas 1996). MPB epidemics may have occurred from time to time in some warmer, low to middle elevation lodgepole pine forests. Some observations suggest that MPB abundance is growing at lower elevations on the northern end of the Snowy Range and in the extreme southeastern part of the Sierra Madre (Jerod DeLay, personal communication).

MPB epidemics typically start in large diameter trees that are diseased or stressed. Most outbreaks occur in stands that are at least 80 years old, with trees at least 8 inches DBH (Koch 1996, Schmid and Mata 1996). Typically, the larger trees are killed in groups of up to 100 (Schmid and Mata 1996). The result is to reduce the average diameter of living trees in a stand while creating openings of varying size (Schmid and Mata 1996). By killing large trees, MPB epidemics increase the number of standing dead trees that can become snags, which eventually fall and become coarse woody debris (CWD), thereby improving habitat quality for a number of species (Balda 1975, Amman et al. 1977). Also, the canopy gaps stimulate the growth of understory plants and small trees (Stone and Wolfe 1996), which creates a ladder of fuel from the forest floor to the canopy as they grow taller. Thereby, flammability increases because of the beetles but no studies have shown thus far that fire frequency or intensity increases. Notably, the total productivity of the forest can return to pre-epidemic levels within 10-15 years (Romme et al. 1986).

The frequency and severity of MPB epidemics vary considerably in the Rocky Mountains, depending on environmental conditions such as elevation and climate. On the scale of an entire national forest, the range of epidemic recurrences is 20 to 40 years, while return intervals for an individual stand are longer (50-100 years) (Johnson 1995, Schmid and Mata 1996). Duration of individual epidemics may vary from one to ten years, with an average of six (Schmid and Mata 1996). Amman et al. (1977) estimated that, at the latitudes of the MBNF, lodgepole pine forests up to 2880 m (9,500 ft) are susceptible to only 25 to 50% mortality, while those above 2880 m (2576 m on the Laramie Peak unit) would likely incur less than 25% mortality.

As with MPB, the spruce beetle (SB) has not been an important cause of disturbances on the MBNF during the last 150 years (von Ahlefeldt and Speas 1996, p. 6-10), though it has caused extensive mortality elsewhere in the region. The SB does exist at endemic levels in the spruce-fir forests of the MBNF, with small outbreaks reported for Green Mountain south of Encampment in the Sierra Madre and Upper French Creek in the Snowy Range. In contrast, much of Colorado and central New Mexico have had severe epidemics that have caused heavy spruce mortality (Baker and Veblen 1990, Veblen et al. 1991, Veblen et al.

1994). The two largest recorded outbreaks in Colorado occurred in the mid- to late-1800s (Sudworth 1900, Baker and Veblen 1990) and in the 1940s (Hinds et al. 1965). SB outbreaks can be associated with accumulated logging debris or stands that have been subjected to windstorms (Schmid and Frye 1977, Veblen 2000).

HRV estimates for spruce beetle epidemics are based primarily on studies and observations from Colorado and New Mexico. The 1940s epidemic, which killed 99% of overstory spruce trees across more than 283,500 ha (700,000 acres) on the White River National Forest, is believed to be an extreme example (Hinds et al. 1965, Schmid and Mata 1996). Historically, it appears that SB epidemics kill anywhere from 5 to 99% of large spruce trees (> 4 in. DBH) over areas ranging from small patches to thousands of hectares (Schmid and Mata 1996). Based on tree-ring records in a part of the White River National Forest, Veblen et al. (1994) estimated that the average return interval and rotation period for SB epidemics was 117 years (range = 111-122 years) and 259 years, respectively, which is more frequent than fire in their study area. Because the SB typically does not attack trees less than 70 years old, epidemics are not likely in stands that have experienced disturbance in less than that amount of time (Veblen et al. 1994).

Susceptibility of spruce stands on the MBNF to future SB epidemics is uncertain. Endemic populations are normally found in windthrown or otherwise fallen spruce trees (i.e., logging residue), and population sizes can increase rapidly after extensive blowdowns (Schmid and Frye 1977). The 1940s epidemic in Colorado, for example, was triggered by an extensive blowdown in 1939 (Hinds et al. 1965). Susceptibility to SB epidemics has not been defined as precisely as for MPB, but Schmid and Frye (1977) suggested the following four characteristics of highly susceptible stands: 1) average DBH >16 in; 2) basal area >150 ft^2 per acre; 3) >65% spruce in the canopy; and 4) locations in well-drained creek bottoms. The effects of a SB epidemic on the MBNF would likely be similar to a MPB epidemic, with the creation of additional snags, more CWD, and a higher level of fuel continuity that could increase flammability while simultaneously improving habitat for other species. Notably, because the SB does not attack subalpine fir, the fir can become more abundant rather quickly after a SB epidemic (Schmid and Hinds 1974).

Numerous other insects are known to occur on the MBNF, such as the western spruce budworm, western balsam bark beetle, pandora moth, pine sawflies, Engelmann spruce weevils and needle miners, but thus far their effects appear to be negligible. Western spruce budworm has been an important cause of disturbance in the mountains of Colorado and New Mexico (Swetnam and Lynch 1993, Veblen 2000). Contrary to its name, the spruce budworm attacks primarily Douglas-fir and white fir, two species that are not common on the MBNF. The spruce budworm has killed some Douglas-fir in the southwestern part of the Snowy Range unit, and western balsam bark beetle is killing groups of 70 or more subalpine fir trees in the Snowy Range and Sierra Madre (Jerod DeLay, personal communication). These insects and others typically cause mortality or low vigor in individual trees or small groups of trees, thereby changing forest structure. The outbreak of the western balsam bark beetle that is just now developing could have been triggered by the drought that has been in effect during the last three years (1999-2002). As with large fires, insect epidemics occur periodically and may be partially caused by unusual climatic conditions (Swetnam and Lynch 1993).

Overall, Schmid and Mata (1996) suggest that insect populations and their effects in the southern Rocky Mountains are currently within their HRV, a conclusion that appears to merit only moderate confidence considering that measurements of epidemics are imprecise. However, it is clear that the insects affecting Rocky Mountain forests are native and probably have reached epidemic population sizes from time to time during the last several thousand years. Human influences such as silvicultural treatments and fire suppression have the potential to move epidemic frequency and severity either below or above historic levels (Keane et al. 2002), but periodic episodes of severe insect damage most likely will always be a part of Rocky Mountain forest dynamics.

4.2.3. Disease

Tree pathogens such as dwarf mistletoe (Arceuthobium americanum), comandra blister rust (Cronartium comandrae), root diseases, broom rusts, and heart rots are also native elements of lodgepole pine and spruce-fir forests (Knight 1994a, Johnson 1995, Veblen 2000), often having the effect of reducing tree growth in a stand or causing localized mortality. Consequently they are commonly viewed as detrimental to "forest health." Despite negative effects on short-term productivity, plant pathogens have been widely recognized for their positive contributions to biological diversity and various ecosystem processes (Gill and Hawksworth 1964, Hawksworth 1975, Tinnen et al. 1982, Dinoor and Eshed 1984, Tinnen 1984, Zimmerman and Laven 1984, Johnson 1995). As with other natural causes of disturbance, the diseases that currently characterize the high-elevation forests of the MBNF probably have always played an important role in long-term ecosystem dynamics. Notably, the introduced white pine blister rust (Cronartium ribicola) is beginning to infect some limber pine trees in the Laramie Mountains and Sherman Mountains units of the MBNF.

The most widespread parasite in high-elevation forests on the MBNF is dwarf mistletoe, a parasitic vascular plant commonly found on lodgepole pine. The effects of dwarf mistletoe on the tree include slowed growth, spike tops, witches brooms, dead branches and eventual tree death (Hawksworth and Wiens 1972). While such effects can have negative consequences for silviculture and the safety of forest visitors, they also may be beneficial for a variety of animals (Hawksworth and Johnson 1989). Bird abundance and diversity were found to be higher in ponderosa pine stands infected with mistletoe in Colorado (Bennetts et al. 1996), and the same could be true for lodgepole pine stands.

Because of its potential to reduce timber yields, dwarf mistletoe has received considerable attention on the MBNF and elsewhere. The mistletoe is not common in the Laramie Mountains, but the full range of infection levels is found on the Snowy Range and Sierra Madre, especially at relatively low, dry sites (Fig. 18; Hawksworth 1958, von Ahlefeldt and Speas 1996, Selmants 2000). In the earliest surveys of the Rocky Mountains, Mason (1915) described the distribution of severe mistletoe damage as scattered and localized. Decades later, Hawksworth (1958) surveyed the MBNF and found infection rates (percentage of trees infected) of 70% in stands that had been tie hacked or selectively cut from the 1860s to 1950. He also surveyed virgin stands and regenerating burns, finding them to be 51% and 27% infected, respectively. In 1977, Johnson et al. (1978) reported that 60% of all lodgepole pine on the MBNF were infected with dwarf mistletoe, but they did not describe the spatial variation in severity levels. Selmants (2000) found that as many as 25% of the lodgepole pine trees in stands developing after clearcutting in the Snowy Range could become infected within 40 years.

Most recently, Kipfmueller and Baker (1998b) surveyed a portion of the Snowy Range and found mistletoe infection in 51% of the stands they examined. Interestingly, they mapped the distribution of mistletoe severity and found that the majority of infected stands had low to moderate infection levels, with heavy infection only in scattered, localized infection centers. Historically, the spread of heavy infection centers was limited most likely by stand-replacing fires. Where such fires kill all infected trees, they essentially eliminate dwarf mistletoe from the stand (Hawksworth and Wiens 1972).

However, the relationship between fire and dwarf mistletoe is complex. Where infected trees survive a fire or timber harvest, the mistletoe can infect new seedlings, creating new infection centers. Notably, some old stands are not infected (Kipfmueller and Baker 1998b). Given the generally slow spread of dwarf mistletoe (less than one meter per year (Johnson 1995), it is uncertain if the effect of fire suppression on infection rates can be detected (Kipfmueller and Baker 1998b). Overall, the severity and distribution of dwarf mistletoe infection on the MBNF is likely within the HRV, with the possible exception of some severely infected areas associated with selective timber harvests that did not remove infected trees.

Two other diseases have received a considerable amount of attention on the MBNF—comandra blister rust and armillaria root disease. Although their life cycles are different, they both cause varying amounts of deformity and mortality to lodgepole pine (Johnson 1995). Lundquist and Lister (1991) found that 27-35% of the stands examined in the Laramie District of the Snowy Range unit were infected with comandra blister rust. This compares to an infection rate of over 50% in the Wind River Mountains (Geils and Jacobi 1984). Comandra blister rust requires specific site conditions (Geils and Jacobi 1991), including proximity to its secondary host plant, Comandra umbellata. Losses from armillaria root disease appear to have been minor on the MBNF (Johnson 1995). Armillaria is most common on Douglas-fir and subalpine fir.

Of the other pathogens present in the high-elevation forests of the MBNF, most seem to have minor effects. All diseases undoubtedly interact with other disturbances and various ecological processes (Zimmerman and Laven 1984). Overall, most stands are likely to have numerous pathogens as concurrent members of the biological community, which helps to ensure the diverse elements of forest structure required for the biological diversity that has existed for millennia (Lundquist 1993, 1995a, 1995b). Except where exotic pathogens such as the white pine blister rust are identified, there is no basis for saying that the current abundance of the native diseases present on the MBNF at higher elevations is beyond the HRV (moderate confidence),

4.2.4. Wind

Because of high elevation, latitude and location, the MBNF is often subject to very high winds, with recorded gusts over 160 km/hr (100 mph) (Marston and Clarendon 1988). Such winds may topple trees in small clumps or over thousands of hectares (Alexander 1964, Veblen et al. 1991). In 1987 a wind storm blew down 6000 hectares of forest in the Teton Wilderness (Knight 1994a), and in 1997 winds of 200-250 km/hr toppled >10,000 ha (24,700 acres) of forest on the western slope of the Routt National Forest in Colorado, just south of the MBNF (Baker et al. 2001; Lindemann and Baker 2001, in press). Two high-elevation blowdowns have been recorded on the MBNF during the past 100 years (von Ahlefeldt and Speas 1996).

Because large blowdowns are relatively rare, their effects are largely unknown. Such disturbances are even more stochastic than fires and an estimation of return intervals is not now possible, nor are estimates of the HRV for the effects of blowdowns. We can safely say, however, that strong winds will continue to be an important factor in shaping landscape patterns and the stand structure of high-elevation forests.

Often, blowdowns interact with other kinds of disturbances. For example, trees weakened by fire, insects and disease may be more susceptible to blowdown; and recently downed trees (Hinds et al. 1965, Schmid and Frye 1977) may trigger spruce beetle epidemics. Flammability surely is increased by a blowdown as well. Wind fall has been a common problem associated with timber harvest, both along clearcut edges (Alexander 1964, Vaillancourt 1995) and in selectively cut stands where a high percentage of the trees are removed (von Ahlefeldt and Speas 1996, page 4-29 and 6-44). Notably, many trees in a stand are not toppled during blowdowns, making the effect more similar to a selective harvest than a clearcut (Lindemann and Baker 2001).

Recent studies of the Routt-Divide blowdown in northcentral Colorado indicate that various features of the physical environment and the vegetation determine where blowdowns are likely to occur. Physical factors seemed to be most influential, especially wind exposure, elevation, and aspect (Lindemann and Baker, in press). Unexpectedly, soil permeability, soil water-holding capacity, and distance to natural edges were not significantly correlated with the blowdown pattern. The effect of cover type varied with elevation. At lower elevations, aspen forests were less susceptible to blowdown and spruce-fir forests were more susceptible. At higher elevations, closer to the Continental Divide, younger stands were less susceptible than older stands (Baker et al. 2001). These results suggest that both topography and vegetation structure can influence the extent and pattern of damage, but that the effects vary with elevation and wind intensity. In another study of the Divide blowdown, Lindemann and Baker (2001) found that blowdown patches were larger and had more compact shapes than patches in the surrounding forest (which were derived from past fires, insect epidemics, topographic influences, and possibly previous blowdowns). They also discussed the potential for a spruce beetle outbreak in the vicinity of the blowdown, concluding that it is difficult to predict the spread of the possible outbreak because it can continue for a decade or more and the beetles can fly long distances (300 m or more in Alaska; Werner and Holsten 1997).

4.3. The effects of timber harvesting on HRV

To some degree, harvesting has effects that are similar to fires and other natural disturbances. For example, canopy gaps of varying sizes are created. However, timber harvesting is different in other ways. Indeed, HRV analyses are motivated to a large degree by the fact that harvesting affects an ecosystem differently than natural disturbances. To illustrate, Muir (1993) found that lodgepole pine in western Montana was likely to be more dominant after a fire than after a clearcut, and that within-stand variability in tree age is less after a fire than after clearcutting. Also, fire tends to leave much of the large wood, while timber harvesting commonly removes it. In the sections that follow, we compare the effects of timber harvesting on stand structure and landscape structure to the effects of natural disturbance regimes.

4.3.1. Stand structure of high-elevation forests

The following attributes or variables can be used to describe stand structure: tree density, percent canopy cover, the density of canopy gaps, the density and cover of understory plants, the ratio of above-ground biomass and below-ground biomass, tree species diversity, total plant diversity, the number of trees in a series of age classes and size classes, forest floor depth, the density of snags, and the amount and distribution of coarse woody debris. Animals and other organisms can have significant effects on stand structure. However, stand structure per se is essentially a function of the vegetation. Small-scale patches or gaps of one to several hundred square meters, such as those created by the death of one or a few trees, are an aspect of stand structure. When discussing landscape structure, larger gaps of one to several hectares are considered. Both stand and landscape structure change through time as well as space because of natural and human-caused influences.

Regrettably, there are essentially no data on stand structure during the reference period (1600-1850). HRV analyses are still possible, but they must be tentative and, as noted, they are based on what is known about how current stand structure changes with time during succession and spatially from one place to another, along with impressions of what the forests must have been like prior to the arrival of European-Americans (based on the journals of early explorers and information about the biology of the dominant species). In the synopsis that follows, we review the history of silvicultural practices on the MBNF and the characteristics of forest stands as they exist today. We also present our conclusions on whether or not past management practices are causing key ecosystem variables to exceed our best understanding of their HRV, indicating our confidence level in drawing each conclusion. This analysis cannot be site specific, but the information that we present can be used when developing site-specific management plans. The focus in this section is on forests dominated by lodgepole pine, Engelmann spruce, and subalpine fir. Aspen groves are considered briefly, but are discussed more fully in Section 6. Ponderosa pine, Douglas-fir and limber pine forests and woodlands are considered in Section 5.

Silvicultural treatments are designed partially, if not entirely, to manipulate stand structure in a way that favors the growth of certain species. Selective harvesting, for example, creates small gaps that are similar to natural gaps caused by the death of a single tree or several neighboring trees. Light penetration to the forest floor is increased and competition for water and nutrients is reduced, thereby enhancing the growth of the remaining trees as well as some understory plants. Selective harvests create or maintain forests with an uneven-aged structure. Over the years, timber harvests also have been prescribed to improve wildlife habitat, such as by increasing the abundance of aspen, or to increase streamflow by catching more snow in small patch cuts where less of the snow water is used by trees.

4.3.2. Silvicultural practices

Until about 1950, selective or partial cutting in lodgepole pine or mixed conifer stands was the predominant form of timber harvest on the MBNF. Native Americans selected individual trees that would meet their needs, such as for lodge poles. Usually they harvested small trees. Impacts from their harvesting were undoubtedly minor compared to those of European-American settlers after the mid-1860s, when trees of 10-14 inches dbh (25-36 cm) were cut for railroad ties (tie-hacking) and telegraph poles. Tie hacking was unregulated and became widespread, especially in the Snowy Range unit (see Fig. 7). Mullison and Lovejoy (1909) reported that the Medicine Bow Mountains yielded 320 million board feet (mmbf) of wood from 1868-1902.

Although selective harvesting is typically not stand-replacing, the majority of lodgepole pine and spruce-fir forests in the Snowy Range and Sierra Madre date back to the mid- to late-1800s (see Figs. 19a and b). The combined effect of selective harvest and fires during this period apparently led to the initiation of many new stands. Young et al. (1989, 1994) found evidence on the MBNF suggesting that riparian areas along streams commonly used for floating railroad ties had also been cleared of most trees. Specifically, they found even-aged, early successional lodgepole pine forests with no large trees or snags along such streams. Along streams that did not have tie drives, the stands were structurally diverse and older, with mixed age and species composition.

During the early years of National Forest administration (1902-1949), methods for harvesting became regulated and gradually evolved toward what they are today. As summarized by von Ahlefeldt and Speas (1996), tree harvesting proceeded as follows:

1902-1914 -- Selective removal of live and dead trees, taking no more than 35% of a stand, with piling and burning of slash to reduce fire hazard.

1914-1922 -- Removal of approximately 35% of the trees 10-15 inches DBH (25-38 cm), with continued slash burning.

1922-1936 -- At least 45% of trees larger than 10 inches DBH (25 cm) were cut, with many larger sawlogs left on the ground; some but not all slash was piled and burned.

1936-1940 -- Only trees that could be used for ties were removed; 65-85% of stand volume could be taken. The demand for railroad ties gradually diminished, with the last tie drive on the MBNF in 1940.

After about 1950, clearcutting became more common because of an increased emphasis on maximizing wood production for construction purposes, plus a desire by some silviculturists to restructure stands that had been tie-hacked or high-graded. In general on National Forests, old forests with comparatively little wood production commonly were clearcut to produce even-aged stands of rapidly growing trees that could be more easily subjected to thinning and other management activities (Langston 1995). The clearcuts were thought to be similar to large fires, one of the kinds of disturbances with which the forest species had evolved. Indeed, both fires and clearcuts do create a patchy landscape mosaic (though with different edge characteristics), and lodgepole pine has proven to be well adapted for establishment in the patches created by both types of disturbances. In some areas in the Rocky Mountains, Engelmann spruce and subalpine fir can be important pioneer species (e.g., in Grand Teton National Park; Doyle et al. 1998). Tree regeneration has been very good on most sites below 2888 m (<9,500 ft) elevation. In fact, lodgepole pine commonly regenerates so abundantly on the MBNF that precommercial thinning is common about 20 years after a clearcut (Grasso 1981, Geils and Jacobi 1991).

In contrast, new tree establishment is much slower near treeline on the MBNF. The last natural disturbance in such areas was probably a fire or windstorm that occurred several centuries previously. Climatic conditions there are marginal for tree growth, especially when high winds blow the snow into downwind forests—killing many of the new seedlings and sapling because of the lack of sufficient snow to protect the plants from winter desiccation and high-intensity solar radiation. Reforestation usually does occur, but more slowly than was expected at the time of harvest.

The 1985 management plan for the MBNF (page III-45) specifies that the rotation age for clearcutting in subalpine/montane landscapes is 90-140 years for stands of lodgepole pine, 80-120 years for stands of aspen, and 100+ years for stands of other cover types. For two- and three-step shelterwood cuts, the rotation ages are 100-180 for stands dominated by Engelmann spruce and subalpine fir, 90-140 years for lodgepole pine, and 100+ for other cover types. If a mean fire-return interval of 150-200 years is assumed for natural disturbances such as fires (Romme and Knight 1981, Romme and Despain 1989), then the silvicultural rotation ages in general tend to be shorter by possibly 10-100 years, depending on the mean fire-return interval at specific sites. Other approaches have been recommended for characterizing fire interval (Kipfmueller and Baker 2000, Baker and Ehle 2000, Howe and Baker, in press), and would give varying results, but fire rotations generally appear to be longer than silvicultural rotations.

The stand age at which harvesting usually occurs also is different from that for natural disturbances. For example, both mountain pine beetle and spruce beetle typically affect stands that are at least 70-80 years old. In contrast, precommercial thinning is often prescribed for 20 or 30 year-old stands (Geils and Jacobi 1991, Koch 1996). Further variation from natural disturbances also has resulted from machine yarding, where whole-trees are brought to the landing before delimbing. This results in a concentration of slash and residual coarse woody debris on the edges of the clearcuts. Following natural disturbances, this material would be distributed more evenly across the affected area. The machinery used for this kind of yarding, as well as for slash treatments such as piling and burning, roller chopping, and tractor walking, can scarify the soil in ways that favor the establishment of lodgepole pine and Englemann spruce seedlings. However, such scarification may also slow the recovery of some understory plants, such as dwarf huckleberry (Vaccinium scoparium), as suggested by von Ahlefeldt and Speas (1996) and the results of Selmants (2000).

As noted previously, HRV analyses should consider the spatial and temporal variability of specific variables. Based on the available information reviewed in previous sections, the nature of silvicultural practices, and our current understanding of ecosystem dynamics, the summary statements that follow seem reasonable for variables pertaining to stand structure in the mid- to high-elevation forests of the MBNF.

4.3.3. Tree density

The naturally occurring number of trees per hectare is highly variable through space and time, ranging from a few hundred mature trees in open stands to over 15,000 in 100-year-old doghair stands of lodgepole pine. Various studies have shown that high lodgepole pine seedling densities are partially a function of the number of serotinous trees in a stand and fire intensity (Franklin and Laven 1989, Anderson and Romme 1991, Turner et al 1997). Ten years following the 1988 fires in YNP forests, tree seedling density ranged from essentially no seedlings to extremely high densities (sometimes exceeding 50 seedlings per square meter). With succession, densely stocked stands are slowly thinned by natural mortality, but many very dense “doghair” stands persist for well over 100 years—probably until the next stand-replacing disturbance. The same occurs in the MBNF. When thinning is done for enhancing tree growth, the goal is to have approximately 1500 trees per hectare (860-1240 trees/ha at age 20 on the Laramie District (Geils and Jacobi 1991)). It is doubtful that timber harvesting creates tree densities in stands that are higher or lower than the HRV (high confidence), but there may be more stands of low tree density in the landscape than during the reference period because of silvicultural thinning (moderate confidence), as discussed in Section 4.4.7. Where thinning has occurred, the trees probably are more evenly distributed than would occur otherwise, simply because even spacing is considered to be advantageous for tree growth.

4.3.4. Seedling and sapling density

Following timber harvesting, fire, or other disturbances, the establishment of new seedlings can be quite low or very high (Turner et al. 1997). Engelmann spruce, subalpine fir, aspen, and lodgepole pine can be pioneer species, though lodgepole pine usually is most common in the early years of succession. Seedling establishment on high-elevation sites with exposure to west and northwest winds is typically slower than elsewhere, regardless of the type of disturbance. Aspen persists in the stands primarily through root sprouting, though seedlings were abundant in some areas after the 1988 fires in YNP (Romme et al. 1995). Spruce and lodgepole pine are thought to prefer mineral soil for seedling establishment, but they often become established where competition for water and nutrients is reduced even if the forest floor remains intact, such as after a wind storm, insect epidemic, a timber harvest with minimal disturbance to the forest floor, or less intense fires. There is no evidence to suggest that seedling and sapling density is beyond the HRV (high confidence). As noted, the density of lodgepole pine seedlings following both fires and clearcutting is often very high (Grasso 1981, Geils and Jacobi 1991).

4.3.5. Percent canopy cover

There is no basis for arguing that processes involved with natural forest development and succession lead to maximizing the rate of tree growth, or even total forest productivity (annual tree growth plus annual understory plant growth). Biologically, plant adaptations facilitate persistence and the production of seed more than growth rates. By thinning forests to maximize tree growth, silviculturists create a canopy that is more open than would have occurred otherwise, thereby pushing percent canopy cover below the range of means for this variable during the HRV period in forests of comparable age and site conditions (moderate confidence). This effect may last no more than about 10 years, as the canopies of the remaining trees normally expand to fill the openings that were created. Notably, canopy cover includes the cover provided by standing dead trees (snags), such as following a fire. This kind of cover provides habitat for some animals and probably is important for the establishment of some plants.

4.3.6. Density and size of canopy gaps

Canopy gaps occur episodically in all forests. If a sufficiently large group of neighboring trees die, perhaps 15-30, a root gap is created as well, where competition for water and nutrients is reduced significantly (Parsons et al. 1994). Gaps, large or small, have a great influence on the growth of understory plants (including saplings), and indirectly they affect age class structure, size class structure, species diversity, and the abundance of some kinds of animals and other organisms. Silvicultural systems that include group selection and thinning could create canopy and root gaps that are similar to those found in unmanaged forests if not too many trees are harvested. However, as with percent canopy cover, the number and size of gaps in a managed forest is likely to be higher than the range of means for this variable during the HRV period (low confidence), simply because the motivation is to enable all remaining trees to grow faster.

4.3.7. Density and cover of understory plants

If managed forests have lower canopy cover or a high density of canopy gaps, the density and cover of understory plants probably will be higher. However, timber harvesting can leave an abundance of slash that could suppress some understory plants, as could the soil compaction, scarification or other changes caused by the machinery that might be used. Species composition can be affected by slash treatments that disrupt the soil profile, which reduces the sprouting capacity of some understory plants and can be favorable for the invasion of new species with wind-borne propagules. Some exotic plants can invade both managed and burned forests, but thus far they apparently do not persist in 30-50-yr-old post-harvest stands at high elevations in the MBNF (Selmants 2000). Exotic plants often are common in road ditches, but they have not yet invaded the adjacent forests in great numbers (Dillon 1998). Given that the density and cover of understory plants is highly variable in natural stands as well as in managed stands, there is no indication, to date, that timber harvesting would push variables associated with the understory vegetation beyond the HRV (moderate confidence). Some plant species may be more sensitive to timber harvesting than to fire, wind storms, or other natural disturbances, and, of course, the road building associated with harvesting decimates the native vegetation and often provides habitat for introduced, weedy species.

4.3.8. Ratio of aboveground biomass to belowground biomass

The root/shoot ratio can vary from about 0.2 to 0.3 in natural lodgepole pine forests, with high values more likely to occur in high tree density stands (Pearson et al. 1984). Silvicultural practices have an effect on tree density, but shifts in root/shoot ratios attributable to harvesting probably are not large enough to exceed the HRV (high confidence).

4.3.9. Tree species and genetic diversity

Apparently all of the tree species present in the 1700s and 1800s are still very common on the MBNF. Initially, high grading through selective cutting may have left an abundance of trees with less desirable wood characteristics, such as subalpine fir, or a large number of malformed or slow growing trees; and more recently the practice of clearcutting may have increased the abundance of lodgepole pine. However, fires may have caused the same shifts. No new tree species have been introduced. Research is insufficient to indicate that harvesting during the last century has changed the genetic diversity of stands, though if fire suppression has been effective, or becomes more effective in the future, the abundance of serotinous lodgepole pine trees in managed forests may decline. Currently, the abundance of stands with at least some serotinous lodgepole pine is high (USFS data; Phil Krueger, personal communication). Therefore, tree species and genetic diversity probably are within the HRV (high confidence).

4.3.10. Total plant species diversity

The species diversity for any group that exists in any ecosystem is determined in part by disturbances. Timber harvesting, with the concomitant removal of CWD and the effects of machinery and slash disposal practices, is a new kind of disturbance to which some species may not be adapted. However, there are no data to indicate that total plant diversity is lower or higher than the HRV (moderate confidence). Some species may be rare and easily extirpated from an area, such as various species of orchids, but evidence for this happening over a large area is not now available (Selmants 2000). Some plants are thought to be interior forest species that could be adversely affected because of depth-of-edge effects associated with road building and patch cutting (Dillon 1998, Baker and Dillon 2000). However, the depth-of-edge effects on such species, beyond road ditches, are difficult to determine and are not well known. The cover of dwarf huckleberry declines in proportion to the intensity of slash treatment, but this low shrub persists in the stand and gradually increases in abundance (Selmants 2000). Post-harvest stands dominated by lodgepole pine at lower elevations appear to recover more rapidly than stands of spruce and fir at higher elevations (Selmants 2000). Many stands have been subjected to at least two entries for the harvesting of many large trees, with the first one occurring around the turn of the century. In such stands, the species that are present today may be the species that can tolerate timber harvesting practices. Additional study is required to determine which species cannot.

4.3.11. Age-class structure of stands

Our current understanding of high-elevation forests suggests that a full range of age-class structures must have existed in the 1700s and 1800s, from even-aged stands of lodgepole pine that had developed after fires in the 1600s and 1700s to older uneven-stands that had not burned for centuries and were dominated primarily by Engelmann spruce and subalpine fir (possibly mixed with lodgepole pine and aspen). As noted previously, a common approach since the mid-1900s has been to convert uneven-aged stands of older forests to even-aged stands of trees that grow comparatively rapidly. However, the variety of age class structures on the MBNF today, at the stand level, probably is within the HRV (moderate confidence). The proportion of the landscape in forests of different ages is discussed in Section 4.4.

4.3.12. Size-class structure of stands

As with age structure, there must have been a great diversity of size-class structures in the high-elevation forests during the reference period, ranging from stands with uniformly small trees to stands with very large trees coexisting with small and mid-sized trees. The size-class structure of stands would change through time and would vary across the landscape. Fires at high elevations prior to the mid-1800s would have killed many of the big trees from time to time, and probably they would have burned over large areas (possibly 25-50% or more of the Snowy Range or Sierra Madre units, though there is still no evidence for such fires on the MBNF; ~36% of YNP was burned in 1988 by either surface or crown fires). Both fire and timber harvest have continued to convert areas with large, old trees to stands of younger and smaller trees, with the important exception that timber harvesting can extend into areas that had not burned for centuries (such as on leeward slopes or in ravines, both of which are thought by some to burn less frequently). In such areas, the size-class structure is probably outside the HRV because insufficient time has elapsed to allow the growth of new large trees to replace those that were harvested (moderate confidence).

Across the rest of the landscape (i.e., where disturbances were more common historically), the variety of size-class structures, at the stand level, is probably within the HRV because both clearcutting and intense fires would create even-aged stands (moderate confidence). Similarly, insect epidemics and partial cutting, where they occurred, could create a structure characteristic of uneven-aged stands. The unnatural size-class structure created by seed tree and shelterwood silvicultural systems is a short-lived variation because the residual trees typically are cut during the next entry, creating an even-aged stand.

With regard to temporal variation in the size-class structure, it is known that lodgepole pine sapling density after both fire and timber harvesting can vary from extremely high in some areas to very low elsewhere (Turner et al. 1997); and that natural thinning of the pine is slow to occur. This is illustrated by the commonly observed “doghair” stands that have great uniformity and can persist for 130 years or more. With time, either a disturbance occurs that changes the size-class structure dramatically or some natural thinning surely must occur. Because this thinning is very slow to occur in stands of lodgepole pine, however, the kind of thinning recommended by forest managers to promote growth of the residual trees could cause changes that are beyond the HRV (high confidence). Notably, size-class structure appears to be more variable spatially than through time, although this must depend on the initial density of the post-fire or post-harvest seedlings. Some stands that have an initial low density become more dense as new trees become established, while high density stands either change hardly at all or become less dense—at least after a century or more.

4.3.13. Forest floor depth

High-elevation forests in the Rocky Mountains are characterized by an abrupt boundary between the mineral soil and the forest floor, which is comprised mostly of fine and coarse litter in varying stages of decomposition. The abrupt boundary is caused by periodic fires that burn the forest floor and expose much of the mineral soil, after which a new forest floor gradually accumulates because of the slow rate of decomposition (Fahey 1983, Fahey and Knight 1986). Because of the absence of earthworms in coniferous forests, the litter does not become well incorporated into the mineral soil. Consequently, just as there is considerable variation in the ages of the forests on the Snowy Range and Sierra Madre units, there is considerable variation in the forest floor depths of stands.

In addition to time since disturbance, variation in forest floor depth is caused by downslope litter movement due to gravity and flowing water, which creates deeper forest floors on lower slopes than on or near ridge tops. Moreover, fallen logs often accumulate detritus on their upper side, and squirrels create middens of cone scales under some trees. Usually the forest floor is 2-6 cm thick, but for a variety of reasons, including tree fall patterns and middens created by squirrels, the HRV is broad, ranging from essentially no forest floor to a depth of several decimeters. Forest managers often recommend that much of the forest floor be scarified or removed, thereby creating a better seed bed for new tree establishment (Alexander 1966, 1987a); or they tend to suppress fires and minimize disturbance to the forest floor, thereby allowing for a continuation of forest floor thickening—at least until the next fire that cannot be controlled. In either case, the range of forest floor depths is within the HRV in stands over most of the landscape, although small areas may be above or below the HRV due to management activities (moderate confidence).

4.3.14. Mineral soil exposed

In general, the forest floor is important for minimizing soil erosion because it increases the rate of infiltration during periods of snowmelt and heavy rainfall. While a natural fire can expose the mineral soil just as a timber harvesting operation can, a notable difference between the two kinds of disturbances is the depth to which the mineral soil under the forest floor is disrupted (Bockheim et al. 1975, Halpern and Franklin 1990, Dion 1998). Scarification by a bulldozer blade or the blades of a roller-chopper disrupts soil structure to a greater depth than does a fire, which normally burns the litter without breaking up the mineral soil (though some localized erosion and soil heating may occur). In the process of mechanical scarification, the surface roots of surviving trees and understory plants are commonly broken, which could lead to higher mortality than would be expected for plants that often survive fires (Selmants 2000). Fires typically burn plant shoots severely, but except in patches where fuels are exceptionally heavy, the soil temperature typically is not hot enough to kill the roots. Thus, most plants, other than the conifers, survive because they are capable of sprouting from underground tissues that are not affected, especially in montane forest (Selmants 2000).

Therefore, mechanized scarification, where it occurs, creates a new kind of disturbance and the HRV for surface soil disruption is exceeded (high confidence). Notably, lodgepole pine and Engelmann spruce seedling densities can be higher if the mineral soil is exposed, whether by fire or mechanical scarification (Alexander 1987a).

4.3.15. Snag density

The importance of standing dead trees (snags) as a component of habitat is widely accepted (Hutto et al. 1992, Hutto 1995, Bull et al. 1997, Parks et al. 1997, Hagan and Grove 1999). Management prescriptions now specify the number of snags that should be left per hectare, as well as the number of live trees that will become the snags of the future (green tree retention). In the absence of such prescriptions, and in stands with a history of timber harvesting, there are fewer snags now than prior to the arrival of European Americans (Harris 1999) because every natural disturbance, with the possible exception of extraordinary wind storms, creates many dead standing trees. Early photographs often show an abundance of snags, giving forests a “salt and pepper” appearance in black and white photos. Native Americans probably used very few standing dead trees, at least those of large size. However, with the advent of thinning, partial cutting, stand improvement, firewood gathering, and wood harvesting in general, the rate at which old snags are replaced with new snags must have been reduced. Past harvesting on the MBNF surely has lowered snag density to below the range of means for this variable in stands where harvesting has occurred (high confidence). Since about 1950 on the Snowy Range and Sierra Madre units, wood harvesting has occurred on approximately 35% and 23% of the forested land, respectively, that was judged suitable for harvesting. Before 1950, harvesting was done in other areas as well.

4.3.16. Coarse woody debris

Natural disturbances do not remove large pieces of wood from forests. Even after an intensive fire, most of the wood remains in the form of dead standing trees (snags), which becomes coarse woody debris (CWD) within two or three decades (Lotan et al. 1985, Tinker 1999, Laudenslayer et al. 2002). The leaves, branches, and smaller wood are consumed by fires, but Tinker and Knight (2000) found that only about 8% of the wood >7.5 cm was burned after an intense fire in Yellowstone National Park. Other estimates for the proportion of CWD consumed by canopy fires range from 12 to 65% (Tinker 1999). With time, the downed wood becomes incorporated into the forest floor (Graham et al. 1994), creating the impression that many 100+-yr-old stands have very little CWD. However, raking the forest floor reveals the remains of many decomposing logs. New CWD debris is added as the larger trees die and fall, one by one during stand development or in large numbers during a windstorm or after the next fire.

Tinker and Knight (2000), using YNP and MBNF data, also concluded that the amount of CWD remaining after several clearcuts is less than after the same number of fires on comparable sites. Moreover, and as might be expected, they found that the time required for CWD to cover 100% of the ground surface area after a fire was much less than after a clearcut, and that this time interval became similar for a clearcut and fire only when slash amounts left on the ground were doubled from average amounts left using current management practices (in other words, 200% of current CWD amounts in slash could keep practices within the HRV on some sites).

Thus, harvesting is fundamentally different from any kind of natural disturbance and produces a forest that is beyond the range of means during the HRV period for both snags and CWD (high confidence), and probably the ecosystem processes associated with structural features dependent on big pieces of wood. USFS guidelines include recommendations for the amount of CWD that should be left during a harvest (Graham et al. 1994), but still, the recommended amounts will be lower after several rotations of timber harvesting than after the same number of fires on comparable sites (Tinker 1999). CWD can be found elsewhere on the landscape, such as in forests withdrawn from harvesting, but the CWD dynamics in specific stands is changed by timber harvesting.

The consequences of such deviations from the HRV are poorly understood, but CWD provides important habitat for some species of fungi and a variety of small mammals and insects, all of which can be important sources of food for vertebrates (Harmon et al. 1986, Hagan and Grove 1999, Laudenslayer et al. 2002); and the organic compounds derived from decomposing wood must have influenced the development of soils on which we now depend for producing wood in the future. CWD is also known to be important for diversifying the structure of streams (Laudenslayer et al. 2002).

4.4. Landscape structure at high elevations

The spatial and temporal variability of ecosystem variables can also be considered at the landscape scale, where, as noted previously, individual stands are the patches in a mosaic that covers square kilometers. Stand-replacing fires, wind storms, some insect epidemics, road construction, and the silvicultural practices of patch-cutting and clearcutting are the primary causes of landscape changes. Such changes are obvious from aerial photographs and satellite images, and they are known to influence the movement of animals, water, nutrients, and sediments across the landscape. The landscape mosaic also can affect the spread of disturbances and subsequent succession (Turner 1980, Knight 1987, Turner et al. 2001), such as after the fires in YNP (Turner et al. 1997).

The mosaic of stands included in a landscape can be described in terms of patchiness, the number and proportions of different kinds of land cover types (landscape diversity), the amount of land in edge and interior forest habitat, the proportion of the landscape in different land cover types and successional stages (or stand ages), the ratio of forest to non-forest land, the rate at which new patch types are formed on a landscape, and various spatial statistics. As with stand structure, the characteristics of landscapes during our reference period (1600-

1850) must be surmised from information on current forests and the dominant species (summarized in Section 3.2).

All available evidence suggests that high-elevation, forested landscapes encountered by early explorers in southcentral Wyoming must have included large areas of even-aged cohorts that originated during crown fires that occurred perhaps every 100 to 200 years or more. Uneven-aged stands of older forest would be found in ravines and other places where fires were less likely to burn or where, by chance, there had not been a fire for several centuries (Romme and Knight 1981). The landscapes probably were patchy due to uneven burning (Foster et al. 1998, Howe and Baker, in press). Large fires would burn for several weeks or months, with greater intensity some days than others. Winds would shift from time to time, causing the flames to move in different directions. Also, spotting would occur and the flames often would move more rapidly uphill than downhill. New edges would be created between burned and unburned stands, creating environments favorable for the windthrow of both dead and live trees as well as the spread of certain insects and pathogens. Moreover, different parts of a burn would experience different successional trajectories, with doghair stands of lodgepole pine in some areas, stands of average or below-average tree density elsewhere, and various combinations of aspen, Engelmann spruce, subalpine fir and lodgepole pine across the landscape (Stahelin 1943, Turner et al. 1997, Reed et al. 1999). The HRV for succession and successional processes surely would have been quite broad.

The only spatially explicit study of pre-European-American landscape patterns on the MBNF thus far was done by Kipfmueller and Baker (2000) on a small, 3,241 ha (8,005 acre) watershed in the northern part of the Snowy Range unit (portions of the Rock Creek and Cooper Creek drainages). Using a detailed analysis of fire history, they reconstructed the landscape pattern as it existed in 1868. They identified 14 stand replacing fires in their study area between 1569 and 1996. Notably, some patches of old forest did not burn during this time. Most of the fires were small, burning less than 200 ha (494 acres). They concluded that climatic conditions played a dominant role in controlling the occurrence of stand-replacing fires, but that fire suppression had been effective in their study area. In a companion report that used the same data set, Baker and Kipfmueller (2001) concluded that the natural fire regime left relatively large, connected patches of interior forest. Ninety percent of their study area burned during the three largest fires, and, as Romme and Despain (1989) found in YNP, there was no evidence for a shifting mosaic steady state on the landscape.

The initial effect of timber harvesting on landscapes, especially clearcutting and the requisite roads, was to "perforate" areas of interior forest. Although edges were created between burned and unburned areas, human-created edges around harvested units and along roads are more abrupt. Moreover, the effects of roads are permanent unless the roads are allowed to revegetate after their utilization for a management activity is completed. The effects of any edge on plants and animals that require interior forest conditions can be detrimental (Murcia 1995, Beauvais 1997, Baker and Dillon 2000); and when depth-of-edge effects become widespread, a landscape is said to be fragmented. In fragmented forests, tracts of interior forest are hard to find, if they exist at all. Large, infrequent disturbances cause fragmentation, whether initiated by humans, fires or wind, even though new forests usually develop in the disturbed areas. (For our report, we define fragmentation as an increase in the number of abrupt transitions between forest and non-forest or young-forest that reduce the area of interior forest and increase the area affected by edges. Some people restrict the use of the term "fragmentation" to disturbances caused by timber harvesting and road building, primarily because the kinds of edges or transitions created by humans are different.) There seems little doubt that the forests of the MBNF are more fragmented now than prior to the advent of large scale timber harvesting and road building (Baker 1994; Baker and Dillon 2000; Reed et al. 1996a, 1996b; Tinker et al. 1998, Knight et al. 2000).

Tinker and Baker (2000) examined the characteristics of clearcut harvesting as it was practiced from 1985 to 1993 on the MBNF. During this short time period, over 95% of scheduled clearcuts under the 1984 forest plan were in lodgepole pine. Typically, multiple clearcuts averaging about six ha (15 acres) were placed within the boundaries of large timber sale areas, with an average of about 550 ha (1,359 acres) that were cut each year. Using these observed patterns, and current MBNF guidelines for clearcut placement (i.e., a minimum age of 90 years and with no new clearcuts placed next to existing clearcuts or stands less than 21 years old), Tinker and Baker used GIS models to project the effects of this harvest regime for 100 years into the future. Results from this simulation showed that the forest would become increasingly more fragmented (i.e., with more edges between old and young forests). Stands >50 ha (124 acres) would be virtually eliminated, as would interior forest; and patch sizes and core area sizes would become much less variable.

Essentially, Tinker and Baker's results suggest that, if practices observed from 1985-1993 continue, the spatially variable landscape patterns created by fire would cease to exist. In their place would be a patchwork of more or less equally sized patches with staggered age distributions. If some old stands are left in tact, all age-classes would continue to be represented on the landscape, but patches of the same age-class would become small and isolated rather than large and contiguous. Tinker and Baker concluded that future landscape structure will have little resemblance to conditions in the 1700s and 1800s if the practices observed from 1985-1993 continue (which included the rather unusual patch-cutting associated with the Coon Creek Project). Other studies of landscape patterns created by clearcut harvesting and road building have reached the same conclusion (Reed et al. 1996a, 1996b; Miller et al. 1996; Tinker et al. 1998; Baker 2000).

The utilization of clearcut harvesting as the primary mechanism for stand-replacing disturbance, rather than fire, leads to several interesting similarities and differences (Fig. 20). First, the amount of area disturbed each year by clearcutting (not including possible edge effects) is actually within the HRV disturbed by fire in YNP, which we believe is a useful proxy for the MBNF because of their similar vegetation. Secondly, however, fire-caused disturbances in most years cover only a small area (0-500 ha), with very large areas (>5,000 ha) affected in occasional years conducive to fire. The fire regime on the MBNF has not changed much in the last 50 years, but fire sizes probably have been much smaller than previously (see Fig. 14) because of successful suppression (Kipfmueller and Baker 2000).

In contrast to fire, clearcutting on the MBNF has tended to disturb a larger and relatively consistent amount of land area each year (mostly 500-3500 ha), plus there is the additional landscape effect of the required road network for harvesting. Therefore, the landscape mosaic created by clearcut harvesting, as practiced from 1950 to 1994, is not likely to resemble the variable mosaic created by a “natural” fire regime. As noted, that has not been a management objective. Notably, on the two units with most of the high-elevation forests (Snowy Range and Sierra Madre), changes attributable to harvesting per se will occur on only about half of the forest land, as that was the amount classified as commercial and suitable for harvesting in 1985 (MBNF Management Plan 1985). If the effects of roads are considered, then the affected portion of the forested land could be 2.5 to 3.5 times larger (Reed et al. 1996a, Baker and Dillon 2000).

The vegetation mosaic of Rocky Mountain landscapes is known to vary greatly through time, primarily because of large-scale fires and other natural disturbances. For example, Romme and Despain (1989) found in YNP that the amount of land area in mature forests occupied about 20% of their study area in the early 1700s, but about 40% in the late 1900s. When the land area occupied by old forests was high, the land area in young forests was low. Such shifts in the landscape mosaic surely have ecological effects on species composition, biodiversity, nutrient cycling, and other ecological phenomena. A similar shifting mosaic must have occurred over much of the high-elevation forests of the MBNF, as suggested by the results of Baker and Kipfmueller (2001).

In the following sections, we summarize our conclusions about specific landscape variables and the probability that they are within or beyond their HRV.

4.4.1. Landscape diversity (number and proportion of land-cover types)

All of the major vegetation types present during the reference period probably are still found today, though in different abundances and with subtle variations that could be caused by climate change as well as by land management practices. Old-growth forests (generally >200 years old with considerable CWD on the ground and a relatively large number of snags) probably cover less area and mid-successional forests (50-150 years old) cover more area, largely due to the impacts of European-Americans since the late 1800s. Humans have caused shifts in the abundance of different cover types and age classes across the landscape, but the number of different kinds of major land cover types probably has not changed (high confidence).

Many of the silvicultural prescriptions during the last 50 years have been for the conversion of slow-growing old forests into fast-growing young forests. The plan in recent years has been to harvest previously harvested forests within a rotation time of about 90-140 years—an interval that is generally shorter than the natural rotations for the kinds of disturbances that occurred in individual, high-elevation stands prior to dominance by European-Americans. Spruce-fir forests may become less common if they are harvested to create pine stands that produce wood more quickly, though some remaining stands of this forest type at high elevations have been classified as unsuitable for harvest. Harvest rotation times of 140 years or less will stop the succession of pine forests to spruce-fir forests; and with fire suppression, aspen may become less common in some areas—though silvicultural prescriptions can favor aspen as well as the different species of conifers. In general, we have the impression that, while changes have occurred, the proportion of high-elevation landscapes in different cover types has not yet exceeded the HRV (moderate confidence), but that this could occur in the future depending on the rate at which spruce-fir forests are harvested and the success of fire suppression. The proportion of the MBNF in different cover types at the present time is summarized in Table 3 and Section 3.2.

4.4.2. Proportion of landscape in different stand-age classes

Studies on high-elevation coniferous forests in YNP suggest that the proportion of land area in young, middle-aged, and old forests has varied considerably during the last several centuries (Romme and Despain 1989). Similar patterns must have occurred in the MBNF. However, a tendency to harvest the larger, slow-growing trees of old forests, and an ability to harvest such forests on north slopes and ravines where fires might not have burned as often, suggests that the amount of old forest today is lower on the MBNF than at any time during the reference period. Approximately 2% of the lodgepole pine and 9% of the spruce-fir forests have been estimated at 200 years old or older on the Snowy Range and Sierra Madre units combined (see Figs. 19a and b). Further study is required to know with confidence if the abundance of old forests is below the HRV for each forest type, but the trend is probably in that direction (moderate confidence). Considering that the proportion of the forests in different age classes fluctuates through time, keeping the proportion of the older forests at a low level would also be beyond the HRV. In YNP, Romme and Despain (1989) found that, since about 1740, at least 10% of the lodgepole pine forests of their study area were >300 years old (LP3 age class), and at least another 15% were 150-300 years old (LP2 age class). Such results further suggest that the MBNF land area in old forests could be below the HRV.

MBNF RIS data suggest that about 25-50% of the forests on the Snowy Range and Sierra Madre units were burned and harvested in the late 1800s and early 1900s (see Fig. 19), creating the large number of 100-125 yr-old stands that exist today. This is reflected in the land area with middle-aged forest (60-150 years old), which appears to be above the HRV (Fig. 21; moderate confidence). Unfortunately, determining the age of older, uneven-aged stands is very difficult, and commonly the results are under-estimates (Kipfmueller and Baker 1998a). Determining if the RIS data are adequate for estimating land area in different age classes during the reference period would require additional fieldwork and is beyond the scope of our study. Therefore, we cannot have a high level of confidence in our conclusion about age-class structure on the landscape. It would be useful to determine the ages of forests when they were harvested or burned during the late 1800s and early 1900s.

4.4.3. Edge, interior forest habitat, and patchiness

Because of increased rates of timber harvesting and road building during the last 50 years, it seems clear, as discussed previously, that the amount of edge has increased and the amount of interior forest has decreased over much of the MBNF (Baker 1994; Baker and Dillon 2000; Reed et al. 1996a, 1996b; Tinker et al. 1998; Tinker and Baker, 2000; Knight et al. 2000, Baker and Kipfmueller 2000). Both characteristics probably have exceeded the HRV for the reference period (high confidence).

A considerable amount of edge during the reference period surely was caused, as it is today, by abrupt environmental changes (Knight and Reiners 2000) and natural disturbances (Veblen 2000), but there are differences in patterns created by human activities. Generally, roads and harvested plots have edges that are more abrupt and the disturbed areas are more symmetrical in shape and location than for natural patterns. Roads are long and narrow, but they influence a large area in the MBNF (see Fig. 10), as do clearcuts (see Fig. 8), and both are significant causes of fragmented landscapes, such as on the MBNF today. Most likely, there are now more mid-size patches (created by clearcut harvests) that are more regularly distributed across the landscape than during our reference period (moderate confidence), when there probably were larger patches of burned forest due to larger fires.

4.4.4. Ratio of forest to non-forest land

Successful regeneration of new stands usually follows timber harvesting on the MBNF, though the rate of tree establishment and growth has been slower on wind-swept sites at high elevations (such as the French Creek clearcuts visible from the Snowy Range road). Usually the stocking rate is very good, if not too high, from a silvicultural perspective. Our observations suggest that harvesting has not enlarged subalpine meadows, nor have trees invaded many of the meadows. Consequently, the ratio of forest to non-forest land (see Table 3) is probably within the HRV on high-elevation landscapes (high confidence).

4.4.5. Rate at which new patches are formed

The 1985 MBNF management plan suggests that the silvicultural rotations for lodgepole pine, mixed conifer/spruce-fir, and aspen forests would be 90-140, 100-180, and 80-120 years, respectively. Considering that the mean crown-fire interval (point scale) probably was 150-200 years in these forest types at high elevations, and that disturbances other than fire would have affected a smaller area than harvesting, it is likely that the rotation is shorter than the HRV for new patch formation by 20-120 years, depending on the forest type and location and the manner in which MFI is estimated (high confidence).

4.4.6. Proportion of land with high snag and CWD densities

Unlike all other natural disturbances, timber harvesting removes significant numbers of large trees, as discussed in sections 4.3.15 and 4.3.16. Large trees are the source for snags and CWD. Considering the large amount of lodgepole pine and Engelmann spruce that have been harvested on the MBNF from the late 1800s to the present, it seems probable that the proportion of land with high snag and CWD densities is lower than the HRV (high confidence).

4.4.7. Proportion of land with high tree density

Silvicultural prescriptions often call for lowering tree density to increase the growth of those that remain. The thinning of trees has not affected a large area on the MBNF, as patch cutting and clearcutting often have been preferred due to less damage by wind and more effective control of dwarf mistletoe. Still, forest thinning in some areas reduces the proportion of the forested land with trees of high density, possibly to the point where this variable is below the HRV (low confidence). Natural thinning certainly does occur, but optimal spacing for maximum tree growth is not characteristic of forest ecosystems. Most thinning has occurred on sites where there is the potential for high tree density, as that is where tree growth is most likely to make the effort worthwhile.

4.5. Summary for high-elevation forests

In summary for high-elevation forests, human-caused disturbances that occurred during the initial period of European-American settlement in the late 1800s left a significant imprint on the MBNF. Still, the HRV for some variables probably was quite broad during the reference period (1600-1850). At the stand scale, the apparent deviations from the HRV are the number of snags, the amount of coarse woody debris, density and size of canopy gaps, age- and size-class structure in some areas, forest floor depth, and the degree to which mineral soil is affected by disturbances. At the landscape scale, deviations from the HRV also appear to be driven by the introduction of timber harvest as the primary disturbance agent, combined with road construction and the successful suppression of some fires. Since the late 1800s, a large amount of forest has been cut on the MBNF. Also, due to a comparatively large amount of accessible terrain and a high road density (miles of road/square mile), a high level of human-caused edge forest has been created, causing a concomitant reduction in the area of interior forest.

Considering that large areas of forest were disturbed by European-Americans in the late 1800s, the species that are present today, over a century later, are those that have been able to survive. There are insufficient data to identify which species have become extinct due to timber harvesting and road construction, but scientists have identified lists of sensitive species. Further research may identify other sensitive species as well as species that are less sensitive than currently thought.

5. THE HRV OF LOW-ELEVATION FORESTS AND WOODLANDS

Forests, woodlands and savannas composed of ponderosa pine, limber pine, aspen, and/or Douglas-fir are common throughout the Rocky Mountains at low-elevations and are often intermingled with foothill shrublands and grasslands.

When the trees grow densely enough to be considered forests, the stands are small and located in ravines or on north slopes. Lodgepole pine occurs in the foothills as well.

In the MBNF, the most common types of low-elevation forests are those dominated by lodgepole pine and ponderosa pine. Lodgepole pine is not usually thought of as a foothill species, but it occurs at the edge of the mountains (such as on the east side of the Snowy Range unit) where the surrounding terrain is too high or dry for other tree species,. The dynamics of lodgepole pine forests at low-elevations are very similar to those at high elevations, with the possible exception that fire may occur more frequently in the foothills. In this section we focus primarily on ponderosa pine forests.

Ponderosa pine in the MBNF is found primarily in the Laramie Mountains (Figs. 22 and 5d). Generally, it occurs in areas that receive relatively high summer precipitation (280-760 mm/year) and where the growing season is relatively warm (Knight 1994a, 1999). Ponderosa pine seedlings appear to be more sensitive to cold temperatures than those of lodgepole pine (Cochran and Berntsen 1973), which may account for why ponderosa pine is rare in the Snowy Range and Sierra Madre (where it is found only in small areas on warm, south-facing slopes at low elevations). Ponderosa pine is able to form extensive stands in the Laramie Mountains unit, where it comprises 51% of the forested area and is most common at elevations of 1,875 to 2,590 m (6,150 to 8,500 ft). Ponderosa pine is less dense on the Sherman Mountains unit, but it makes up 47% of the relatively small area with trees (38% forested; see Table 3 and Fig. 5c).

Douglas-fir occurs at similar elevations as ponderosa pine, though it grows more densely on mesic sites at somewhat higher elevations or on north-facing slopes. Douglas-fir can be found in small stands in the MBNF (Fig. 23), such as on Sheep Mountain and along the North Platte River, but it is less common than to the west and northwest in Wyoming, in Colorado to the south, and in Montana to the north. On the MBNF, Douglas-fir occupies 2%, 2%, 4%, and 1% of the forested land on the Sierra Madre, Snowy Range, Sherman Mountains and Laramie Mountains units, respectively (von Ahlefeldt and Speas 1996). Despite its low abundance in the MBNF at the present time, some investigators have speculated that Douglas-fir was once more widespread in portions of the Snowy Range, Sherman Mountains, and Laramie Peak units (Douglas 1913, Alexander et al. 1986, von Ahlefeldt and Speas 1996).

Limber pine may occur with ponderosa pine or Douglas-fir, but it is ecologically distinct in various ways. First, it tolerates a wide range of environmental conditions (Fig. 24), as is evident by the fact that it occurs on escarpments at 1528 m (5042 ft) elevation near Pine Bluffs, Wyoming (on the western Great Plains near the Nebraska border) and at alpine treeline above 3,000 m (9.900 ft) (McNaughton 1984, Schuster et al. 1994). Perhaps due to an inability of seedlings to compete with other species, it is often found on rocky soils and ridges (Lepper 1974, Alexander 1986) where other species are not able to survive. Also, the distribution of limber pine appears to be affected by the behavior of the Clark's nutcracker (Tomback 1983), with the birds planting the seeds on exposed ridges and slopes where snow is less likely to accumulate. Based on the RIS database, limber pine occurs on <1% of land area on the Snowy Range and Sierra Madre units, but makes up 2% and 9% of forested land on the Sherman Mountains and Laramie Mountains units, respectively (von Ahlefeldt and Speas 1996).

Depending on various physical and biological factors, ponderosa pine, Douglas-fir and limber pine can occur individually or in mixed stands. Due to the underlying soils and geology, or variations in topography and microclimate, these species often form open woodlands or patchy mosaics, with groups of trees interspersed with shrubs, grasses and forbs. Common species associated with such woodlands in Wyoming include skunkbush sumac (Rhus trilobata), common chokecherry (Prunus virginiana), mountain big sagebrush (Artemisia tridentata var. vaseyana), Oregon grape (Mahonia repens), western snowberry (Symphoricarpos occidentalis), Rocky Mountain juniper (Juniperus scopulorum), ground juniper (Juniperus communis), Idaho fescue (Festuca idahoensis), king spikefescue (Leucopoa kingii), sideoats gramma (Bouteloua curtipendula), and little bluestem (Andropogon scoparius) (Knight 1994a).

Maximum ages for trees in low-elevation forests on the MBNF usually have been estimated at 250-350 years for ponderosa pine, 200-300 years for Douglas-fir, and 500+ years for limber pine. Notably, Brown et al. (2000) found ponderosa pine that are 500 years old in the Laramie Peak area. Stand ages might have been considerably younger or older than the ages of individual trees, but ponderosa pine and Douglas-fir have thick bark, enabling them to survive surface fires, and often they occur on sites that are not highly flammable, such as rocky escarpments (States 1968). Both insect epidemics and fire would have killed some of the trees (usually not all of them), thereby contributing to the open nature of the stands.

5.1. Disturbances at low elevations

While fire is thought to be the predominant factor in the maintenance of overall stand and landscape structure, other disturbances have also played important roles. Mountain pine beetles, for example, have at times caused large stand-replacing disturbances, such as in the Laramie Peak unit (1988-94). Other agents seem to be individually less important— including root rots, wind storms, and heavy frosts as well as other insects—but the interactive, often synergistic relationships between the various disturbance agents present in low-elevation forests have enhanced the natural diversity (Fig. 25; Lundquist 1995a, 1995b).

5.1.1. Fire

Various explanations have been offered to describe ponderosa pine dynamics prior to the time of fire suppression (Fig. 26; e.g., Covington and Moore 1994, Shinneman and Baker 1997, Veblen et al. 2000). The first explanation assumes relatively frequent, low-intensity surface fires that kill most of the small trees but few of the older, thick-barked trees—thus maintaining low-density ponderosa pine woodlands. Historic accounts describing "open and park-like" stands with understories rich in grasses were common in the American Southwest (Cooper 1960); and on the eastern slope of the Laramie Mountains, many early explorers described ponderosa pine as "thinly scattered", "stunted", "scraggy", and growing "most sparsely" (Dorn 1986, pp. 3-8). Photographs taken in the 1800s also documented open stands of ponderosa pine and Douglas-fir throughout the Rocky Mountains (Progulske 1974; Gruell 1983b; Veblen and Lorenz 1986, 1991; Johnson 1987; Parrish et al. 1996).

Fire history studies for portions of the Black Hills and Front Range have generally estimated mean fire intervals (MFI) of 15-45 years (Laven et al. 1980; Fisher et al. 1987; Goldblum and Veblen 1992; Brown 1994; Brown and Sieg 1996, 1999), compared to MFIs of less than 10 years in the Southwest (Weaver 1951, Dieterich 1980, Fule et al. 1997). Ponderosa pine fire scars from a natural area near Laramie Peak (the Ashenfelder Basin) indicate a median fire interval of 26 to 34 years (Brown et al., 2000; Fig. 27). Where Douglas-fir and ponderosa pine stands occur near each other in Colorado, fire intervals for stands of Douglas-fir appear to be slightly longer (Goldblum and Veblen 1992), possibly because Douglas-fir usually occurs on more mesic sites. In a thorough review of the literature for the Colorado Front Range, Veblen (2000) found a range in MFI of 5-40 years. Baker and Ehle (2001) concluded that such ranges are too narrow and are biased toward shorter intervals, but frequent, low-intensity fires probably characterized some low-elevation forests in the Rocky Mountain region.

A second explanation of ponderosa pine dynamics suggests that infrequent crown fires are more characteristic of the ponderosa pine forests than frequent surface fires, such as Shinneman and Baker (1997) proposed for the Black Hills of South Dakota and northeastern Wyoming. According to Shinneman and Baker, disturbances in the ponderosa pine forests of this area were infrequent and unpredictable stand-replacing fires that created large areas of dense, closed-canopy, even-aged forest similar to stands of lodgepole pine. They review the journals and photographs of early explorers such as Dodge (1876), Newton and Jenny (1880), and Graves (1899), who provide evidence that broad expanses of dense ponderosa pine forests and extensive, stand-replacing fires were common in the Black Hills prior to the arrival of European-Americans. Early observations on Casper Mountain, on the north end of the Laramie Mountains (though outside the MBNF boundary), also described "extensive tracts of Pitch Pine [ponderosa]" and "Gigantic pines in great density" (Dorn 1986, pp. 10-12).

In reality, both ponderosa pine savannas and dense forests could have coexisted at low elevations in the MBNF during the reference period, especially in the Laramie Mountains unit. Variability in site conditions throughout the montane and foothill zone likely resulted in variability in fire regimes as well (Thomas Veblen, personal communication). Studies in the Colorado Front Range have identified mixed-severity fire regimes, which have a combination of surface and stand-replacing fires over a period of time in the same area (Shinneman and Baker 1997, Kaufmann et al. 2000, Veblen et al. 2000, Ehle 2001, Ehle and Baker, in press).

Because there is evidence for regimes involving surface, crown, and mixed- severity fires, fire frequency in ponderosa pine and Douglas-fir probably would not have been the same in all areas. This is reflected in part by the significant variation in MFIs among different studies and locations within the region (Table 6). MFI estimates vary widely (e.g., 3-161 years, Laven et al. 1980; 8-82 years for Ashenfelder Basin, Brown et al., 2000), partly because of differences in methodology (Veblen 2000) and partly due to differences in climate, fuel conditions, topographic position, and various uncertainties and biases (Baker and Ehle 2001). Nevertheless, warmer and drier sites (or years) probably experienced more frequent fires, while fires may have been less frequent when or where conditions were more mesic. Veblen et al. (2000) conducted a detailed study of tree rings in ponderosa pine forests of the Colorado Front Range, finding that the alternation of wet and dry periods within a 5-year period is conducive to fire spread. Wet periods apparently associated with El Nino conditions allowed the production of fine fuels that increased flammability during dry periods associated with La Nina conditions. The effect of El Niño, however, may not be as strong in areas north of the Colorado Front Range, such as in Wyoming (Veblen 2000).

A factor complicating our ability to interpret the HRV for fire in low-elevation forests and woodlands is the role that humans have had in influencing the fire regime. Prior to the arrival of European-Americans, fires set by Native Americans probably were most common at lower elevations (e.g., Barrett and Arno 1982, Gruell 1983a) and were inseparable from "natural" fires (Martin and Sapsis 1991). However, in a recent review of the effects of Native Americans on fires in the Rocky Mountains, Baker (2002) concluded that—away from heavy use areas—there is no evidence to suggest that fire regimes were strongly influenced by people until the arrival of European-Americans.

During the past century, however, human influences appear to have lengthened the fire return interval in many ponderosa pine forests throughout the Rocky Mountains (Covington and Moore 1994; Parrish et al. 1996, Veblen 2000; Brown et al. 2000, Keane et al. 2002). On the MBNF, Brown et al. (2000) estimated that fire had not occurred since 1911 and 1909 at two sites within the Ashenfelder Basin (north of Laramie Peak) (see Fig. 27). The resultant 86- and 88-year intervals since the last fire are 12 and 6 years longer than any recorded in the 1800s, and they are over three times longer than the median before the arrival of European-Americans. Notably, Baker and Ehle (in press) argue that additional data are required before such conclusions can be accepted with confidence.

In many areas, it seems clear that fire suppression during the 20th century has contributed to the establishment of dense stands of smaller trees that are now very susceptible to stand-replacing fires (Fig. 26; Covington and Moore 1994, Parrish et al. 1996, Fule et al. 1997, Keane et al. 2002). Livestock grazing also may have played a role by reducing competition for young seedlings from grasses and forbs and reducing the amount of fine fuels, as observed in Utah (Madany and West 1983, Keane et al. 2002). Moreover, favorable climatic conditions during the early 1900s (especially around 1920) could have contributed to higher ponderosa pine densities (Savage et al. 1996), and if adequate moisture was available, warming during the last century could promote ponderosa pine seedling establishment. As a result, intense stand-replacing fires in ponderosa pine forests, in the West as a whole, may be more common presently than during the 18th and 19th centuries (Agee 1997, Keane et al. 2002).

Fire severity, in general, is probably directly proportional to the length of the fire free interval, with more severe fires occurring where or when fire intervals are longer. Fire sizes in stands dominated by ponderosa pine and Douglas-fir probably have been inversely proportional to fire frequency, as in high-elevation forests. At nearby Devil's Tower National Monument, small fires caused by individual lightning strikes were common historically (Fisher et al. 1987). Such fires would typically damage only one or two trees before burning out due to weather conditions or fuel discontinuity. Larger fires were less frequent, occurring only at times when weather and fuel conditions facilitated fire spread. In parts of the Black Hills, where fuel accumulated more rapidly in ponderosa pine forests, fires were larger and more intense, as in lodgepole pine forests (Shinneman and Baker 1997). Historic fires as large as 20,000 to 60,000 ha. (49,400 to 148,200 acres) have been reported in the Black Hills (Dodge 1876; Shinneman and Baker 1997, based on Graves 1899). Since 1890 most fires in the Black Hills have been comparitively small, except for the Jasper Fire in 2002 that burned 33,850 ha (83,580 acres). Notably, many ponderosa pine woodlands and savannas in the MBNF are patchy. This could have limited the spread of fire to smaller areas than in the Black Hills, unless the trees were separated by flammable grasslands.

Since 1945 small fires have been much more common in the Laramie Mountains than more intense, widespread fires (Fig. 28). From 1945-1993, 95% of all fires in the Laramie and Sherman Mountains units were less than 4 ha, while only 11 fires (ca. 1.5%) were larger than 125 ha (309 acres) and only two (0.2%) were larger than 400 ha (988 acres; von Ahlefeldt and Speas 1996). The total area burned on both units from 1945 to 1993 was only 6,190 ha, or 3% of the land area (von Ahlefeldt and Speas 1996). Notably, despite the relatively small extent of fires during the last 50 years, lightning ignitions have been higher on the Laramie and Sherman Mountains than elsewhere on the MBNF (Fig. 29), and two very large crown fires occurred in 2002 in the Laramie Peak area—the Hentzel Fire that burned 6,016 ha (14,855 acres) and the Reese Fire that burned 7,633 ha (18,848 acres).

An examination of the RIS database for stand ages suggests that most of the stands of ponderosa pine in the Laramie Peak area originated about 80-130 years ago (see Fig. 19d), though there are some forests that are considerably older. As with high-elevation forests, these young ages could be attributable to frequent fires and timber harvesting in the late 1800s and early 1900s. However, because of more frequent surface fires and the patchy, multiple-aged nature of many ponderosa pine stands, it is more difficult to determine stand age than for lodgepole pine stands. Such data must be interpreted cautiously, as the stands could be considerably older than Fig. 19d indicates.

5.1.2. Insects

The primary insect affecting low-elevation forests on the MBNF, particularly those dominated by ponderosa pine, is the native mountain pine beetle (MPB; Dendroctonus ponderosae). The native Pandora moth, Douglas-fir beetle, and western spruce budworm may be present also, but their effects are relatively minor (Schmid and Mata 1996). As with lodgepole pine, the MPB survives within the phloem underneath the bark throughout nearly all of its life cycle (McCambridge et al. 1979). Healthy trees can often defend themselves by producing resin to expel the beetle, but stressed or damaged trees may lack this ability.

The effects of MPB epidemics in ponderosa pine are similar to those for lodgepole pine. Epidemics often start when the beetles kill small groups of the larger trees (Schmid and Mata 1996). If sufficient stresses are present (e.g., old age, water stress, or disease), the epidemic may spread across the landscape, killing groups of as many as 100 trees surrounding the stressed "focal trees" (Schmid and Mata 1996). The results of such outbreaks include a reduction in the average tree diameter in a stand and the creation of small canopy gaps as well as large openings (Schmid and Mata 1996). As beetle-killed trees become snags and downed logs, they contribute to structural diversity (and therefore habitat diversity), at the same time providing fuel for the next fire (Lundquist 1995a, 1995b; Schmid and Mata 1996). Other species not susceptible to MPB, such as Douglas-fir, may become more abundant following an epidemic (Schmid and Mata 1996).

Historically, MPB epidemics in the southern Rocky Mountains, including the MBNF, are thought to have been more severe and more frequent in ponderosa pine than lodgepole pine, perhaps because of warmer winters at lower elevations. On the scale of landscapes, an epidemic may occur in some areas every 11 to 20 years (Schmid and Mata 1996), often coinciding with periods of dry climatic conditions (Beal 1943). Return intervals for an individual stand must have been longer, with epidemics occurring once every 50 to 100 years (Johnson 1995, Schmid and Mata 1996). Such frequent epidemics may have contributed to maintaining an open stand structure with low tree density, as even the most severe epidemics probably did not kill all the trees. The duration of an epidemic apparently varied from 2 to 14 years (Schmid and Mata 1996). The severity also varies considerably (McCambridge et al. 1979). In the Black Hills, historical reports indicate that most trees on a 50,000 ha (123,500 acres) tract of forest were killed by a MPB epidemic in 1895 (Shinneman and Baker 1997).

Ponderosa pine has likely evolved with the influence of the MPB, as it has with the influence of fire. However, the increase in tree densities that have been reported can contribute to both competition and water stress, thereby reducing tree vigor and potentially increasing the susceptibility of some stands to MPB. The most recent epidemic on the MBNF occurred from 1988 to 1994 and caused severe mortality at lower elevations near Laramie Peak from north of Esterbrook to Palmer Canyon (Howard Pickerd, personal communication). Because of the apparently unusual tree densities, it is possible that the severity of this epidemic was outside the HRV. It is also possible, however, that the warm and dry conditions of the 1980s were the primary cause. If that is true, then this most recent epidemic may have been a natural event, well within the HRV (unless the warming was human induced, which is difficult to determine if not impossible).

Overall, Schmid and Mata (1996) characterize insect populations and their effects in the Southern Rocky Mountains as potentially within their HRV, but with some important caveats. Specifically, they stress that measurements of "epidemics" are imprecise, which makes comparison of past and present epidemics difficult. It is clear, however, that the insects affecting Rocky Mountain forests are native and have most likely experienced periodic population increases to epidemic levels for thousands of years. Also, while human influences such as silvicultural treatments and fire suppression have the potential to move epidemic frequency and severity either below or above historic levels, insects will always be part of low-elevation forests and woodlands in the Rocky Mountains (Schmid and Mata 1996).

5.1.3. Disease

Various diseases are present in the low-elevation forests and woodlands of the MBNF. The most common include Armillaria root disease (Armillaria mellea), red rot (Dichomitus squalens), Schweinitzii root rot (Phaeolus schweinitzii) and limb rust (Peridermium filamentosum) (Alexander 1986; Lundquist 1995a, 1995b). While such diseases can cause localized mortality and tree damage, none of them, individually, appears to have a great impact on the forest as a whole. Rather, it is the combined effects of diseases and other disturbance agents that influence stand structure and composition (see Fig. 25). In some cases, timber harvest and bark beetle epidemics may enhance the effects of Armillaria root disease and some other parasites (Lundquist 1995a), but these agents have not been reported as serious problems on the MBNF (von Ahlefeldt and Speas 1996). Notably, dwarf mistletoe (Arceuthobium vaginatum) is not common on ponderosa pine in the MBNF, though it is common to the south in Colorado. The rarity of dwarf mistletoe on the ponderosa pine of Wyoming may be due to geographic isolation or factors that either make the trees more resistant to mistletoe infection or provide an environment less suitable for mistletoe (Hawksworth and Wiens 1972).

With one known exception, the diseases found on trees at low elevations are native and are not thought to have been greatly affected by human influences. The exception is the white pine blister rust (Cronartium ribicola), which infects five-needle pines such as limber pine. This exotic disease appears to be migrating southward from the Northern Rockies, where it is a serious problem for white bark pine (Pinus albicaulis). The rust has been found on limber pine in the Pole Mountain unit and in northern Colorado (P. Krueger, personal communication). It probably occurs in the Laramie Peak area as well. The presence of an exotic pathogen could push some ecosystem variables beyond their HRV if its effects and periodicity are different than native pathogens. Unfortunately, such differences are not usually known at the time of invasion, and after that, the elimination of the pathogen is exceedingly difficult, if it's possible at all.

5.1.4. Wind

Depending on the presence of predisposing factors, such as disease, insects, fire, and a shallow root system, the effects of wind will vary from localized tree damage to blowdowns over large areas (see Fig. 25; Alexander 1986, Lundquist 1995a, Shinneman and Baker 1997). Limber pine often occurs on very windy sites, but its "limber" branches are surely an adaptation to reduce breakage. In the Black Hills, Graves (1899) found evidence for one severe wind storm in the 19th century. Another occurred in 1940 (Shinneman and Baker 1997), which leveled 1.5 million board feet of ponderosa pine. Despite the severe effects of such stochastic events (Lindemann and Baker 2001, 2002; Baker et al. 2002), wind is a natural phenomenon that will always be a potential disturbance agent in low-elevation forests and woodlands.

5.1.5. Timber harvest

During the 1860s, the low-elevation forests of the Sherman Mountains unit east of Laramie were harvested extensively for railroad ties, construction materials and fuelwood; and in the Laramie Mountains unit to the north, many trees were harvested for telegraph poles (Grasso et al. 1981, Boglioli et al. 2002). More recent harvesting at low elevations has been limited primarily to ponderosa pine, especially since about 1980 when salvage logging following mountain pine beetle damage was done in the Albany Peak, Black Mountain, Laramie Peak, and Friend Park areas (von Ahlefeldt and Speas 1996). The primary method employed has been thinning (i.e., selective harvest or "beetle harvests"), which was intended to slow the spread of the beetles while providing a supply of wood. Because of the limited extent of such harvests, the land area harvested per year has not been calculated and specific comparisons to other disturbance regimes are not now possible.

In general, one of the primary effects of past harvesting in ponderosa pine has been the loss of large, old trees—sometimes thought of as "overmature." Both selective harvest methods and salvage logging have tended to remove the old trees that were important components of ponderosa pine landscapes (Covington and Moore 1994, Shinneman and Baker 1997). Such trees were a valuable source of wood, but, as noted for high-elevation landscapes, they also provided important elements of wildlife habitat, both as live trees and as snags or downed woody material (Cunningham et al. 1980, Hutto et al. 1992, Crompton 1994, Hutto 1995, Dykstra 1996, Harris 1999). At the same time, the decaying wood provided a mechanism for nutrient storage and a potentially valuable source of organic compounds for the soil.

Another effect of harvesting has been to reduce the natural variability in stand structure and age distribution caused by historic disturbances. In the Front Range of Colorado, harvesting done for the purpose of improving tree growth also produces a greater uniformity in density and age structure (Myers 1974, Alexander 1986). The effects of such an approach can be seen on the Black Hills National Forest, where harvesting by industry and firewood cutters has left fewer snags and less downed wood than in unmanaged stands (Lundquist 1995a, 1995b).

Although past timber harvesting methods may have created conditions that are outside the HRV, new approaches for harvesting can be a means to restore some stand characteristics (Covington and Moore 1994, Romme et al. 2000). Where open stands of large ponderosa pine were common prior to the arrival of European-Americans, the thinning of young trees that have invaded in recent decades, combined with the retention of most larger trees and the application of prescribed fire, has become a common prescription (Harrington 1981, Romme et al. 2000, Kaufmann et al. 2000, Arno and Allison-Bunnell 2002).

5.1.6. Livestock Grazing

Livestock grazing has been a common land use in the MBNF at both high and low elevations. Where the trees are not dense, the forage is nearly continuous. Domestic livestock were first introduced in southeast Wyoming in the late-1800s, but the animals were not regulated on federal lands until the establishment of Forest Reserves around 1900 (Boglioli et al. 2002). Although records are not available before 1925 for the Sherman Mountains unit, and before 1935 for the Laramie Mountains unit, grazing intensity in these areas likely peaked during the early 1900s (von Ahlefeldt and Speas 1996; Robert Mountain, personal communication). For the MBNF as a whole, early reports of the range condition in the Snowy Range and Sierra Madre Mountains indicated overuse and severe erosion (Blackhall 1909, Mullison 1909). A thorough review of the effects of grazing is beyond the scope of this report, but the number of livestock on the MBNF has declined steadily in recent decades (see Section 3.3). Unfortunately, little information on the abundance and year-to-year variability of bison and elk grazing during the reference period is available.

The composition of herbaceous species in ponderosa pine savannas of the early to middle 1800s is difficult to determine. As noted, grazing and fire suppression have been widespread in ponderosa pine savannas, and unaffected reference areas are rare. Likewise, although some historic descriptions of range conditions exist, their usefulness is limited. For example, Douglas (1912) provided detailed information on the various "grazing types" found on the MBNF, but his observations were made after some of the most intensive livestock grazing had already occurred. Johnson (1987) compared historic and modern photographs of rangeland in the lowlands of southeast and south-central Wyoming, revealing that very little visual change has occurred in many areas. Unfortunately, changes in grass and forb species are difficult to detect from photographs. Moreover, some of the photographs themselves, from the 1870s, may have been taken after a period of heavy grazing.

Regionally, some effects of domestic livestock grazing in ponderosa pine ecosystems include the following: 1) The creation of patches from feeding, trampling, and nutrient input (urine and feces); 2) increases in soil erosion where more than 50% of herbaceous cover is removed (Currie 1975); 3) declines in some native species of bunchgrass that are sensitive to grazing (e.g., Johnson 1956, Madany and West 1983); 4) increases in the abundance of grazing-resistant species; 5) the introduction of exotic species such as crested wheatgrass and smooth brome; and 6) increases in tree and shrub density by reducing flammability and less competition from grasses and forbs through herbivory (Madany and West 1983).

Aside from direct effects on vegetation, any discussion of grazing in the context of HRV should consider the populations and behaviors of both native and domestic herbivores before and after the changes brought about by European-Americans. In the ponderosa pine forests and woodlands of the MBNF, at low elevations, we know from historical accounts that native herbivores such as bison, elk, deer and pronghorn antelope were common in the early 1800s (Dorn 1986). It's reasonable to assume that native grasses and forbs should be more resistant to grazing effects than in areas where native herbivores were less common (e.g., the Great Basin deserts to the west; Mack and Thompson 1982). However, because population estimates of native herbivores are not available for the reference period, direct comparisons to modern levels are impossible. Notably, domestic and native herbivores do have different foraging habits. For example, bison typically roam widely, thus distributing their effects over large areas, while cattle often stay close to water, concentrating their effects in a smaller area (van Vuren 1982).

Overall, we don’t know if current grazing in ponderosa pine ecosystems on the MBNF is within the HRV. In 1994, grazing on the entire MBNF was reported at 46,619 AUMs for sheep and 35,223 AUMs for cattle (von Ahlefeldt and Speas

1996). Such numbers are much lower than 80 or 100 years ago, when the heaviest grazing by domestic animals occurred (see section 3.3).

5.2. Stand Structure at low-elevations

Based on written and photographic records from the 1800s, as well as reconstructions from recent studies, qualitative comparisons can be made to the conditions that have developed after more than a century of European-American influence. In this section we review the available information and draw conclusions pertinent to the HRV. In addition to timber harvesting, the effects of fire suppression, livestock grazing and climate change are considered.

5.2.1. Tree density, sapling density, canopy cover, age-class structure, and size-class structure

Qualitative information suggests that low-density ponderosa pine savannas occurred in Wyoming and Colorado during the 1800s (e.g., Progulske 1974, Dorn 1986, Veblen and Lorenz 1991). Moreover, reconstructions from fire scars and other sources indicate that some stands in the Southwest had been maintained for hundreds of years by frequent surface fires (White 1985, Covington and Moore 1994, and others). The same probably occurred at low elevations in the MBNF. However, "dense" stands were found in the Black Hills in the 1800s (Dodge 1876, Newton and Jenny 1880, Graves 1899), probably on more mesic sites, and such stands probably were characterized by infrequent, stand-replacing fires (Shinneman and Baker 1997). The pre-1900 canopy cover might have been 50 to 80% in such areas (Graves 1899). Working in the Black Hills, McAdams (1995) estimated that pre-European tree densities ranged widely, from 15 to 1600 trees/ha. Similarly, Ehle and Baker (in press) reported ponderosa pine densities of 70 to 3,000 trees/ha for Rocky Mountain National Park to the south of the MBNF.

Where frequent surface fires occurred historically, large, older trees >14 inches (35.5 cm) DBH typically survived and dominated most stands. In contrast, many ponderosa pine forests today are dominated by young saplings and mid-sized trees (Covington and Moore 1994, von Ahlefeldt and Speas 1996, Keane et al. 2002). We found no evidence to suggest that shifts in tree species composition have occurred on the MBNF, such as in the Southwest where Fule et al. (1997) found that trees less adapted to fire than ponderosa pine have become more important in many stands (e.g., Douglas-fir and Gambel oak). Increased conifer density has resulted in increased leaf area index (LAI, the amount of leaf area per unit of ground area), which can lead to higher evapotranspiration and a reduction in streamflow (Orr 1975, Covington and Moore 1994, Keane et al. 2002). Effects of this nature might occur in the Laramie Mountains unit, but the proportion of the landscape with this condition is not known.

In general, it seems reasonable to conclude that tree density, sapling density, canopy cover, age-class structure, and size-class structure are outside their respective range of means during the HRV reference period at low elevations on the MBNF, especially on relatively dry sites where fire suppression has been effective (moderate confidence). Even in areas characterized by a stand-replacing fire regime, the flammability may now be higher than normal because of the rapid growth of trees on the mesic sites where such stands are found. Notably, some research suggests that the general trend in tree density with stand age is downward rather than upward in some areas, due to mortality caused by factors other than fire, e.g., in Rocky Mountain National Park (Ehle 2001, Ehle and Baker, in press). This would occur in areas where a dense, even-aged stand of ponderosa pine develops after a stand-replacing fire—similar to those sometimes formed by lodgepole pine at higher elevations.

5.2.2. Spatial distribution of trees

An important pre-1900 characteristic of ponderosa pine forests appears to be the clumped distribution of tree cohorts in some areas at low elevations. Historical observations and reconstructions of open-grown ponderosa pine in the Southwest indicate that trees grew in clumps or patches of less than 0.4 ha (1 acre) and 3 to 44 trees (Cooper 1960, White 1985, Moore et al. 1999). Covington and Moore (1994) found that age structure within and between these clumps varied depending upon scale. Small, even-aged groups of 2-5 trees occurred in a stand that was uneven-aged as a whole (White 1985). Such patterns could occur in the Laramie Peak area, though Ehle and Baker (in press) found that dense, even-aged stands were common in Rocky Mountain National Park. Further research is necessary to determine if ponderosa pine distribution patterns observed in the Southwest are also found to the north.

5.2.3. Snags and coarse woody debris

Working in the ponderosa pine forests of the Black Hills, which has a climate similar to the Laramie Peak unit, Lundquist (1995a) identified ten classes of snags and five classes of logs, finding that different types of disturbance create different types of snags and logs. For example, the stems of beetle-killed trees tend to break, leaving a relatively short snag, while trees killed by root rot are commonly uprooted and become CWD on the forest floor. Timber harvest, in the small amount of area where it occurs at low elevations on the MBNF, tends to 1) reduce the number and diversity of snag types and 2) distribute log types among the five classes more evenly on the forest floor (Lundquist 1995a, 1995b). This evidence, combined with old photographs in Wyoming that show a much larger number of dead trees in low-elevation forests than occurs today, suggests that snags and downed wood are now lower than the HRV (moderate confidence) where harvesting has occurred (including casual fire-wood cutting). Notably, fires in low-elevation forests burn some CWD, possibly more than observed by Tinker (1999) in high-elevation forests because of frequent surface fires. However, with less frequent fires this wood could persist for a longer time, and if CWD inputs remain constant, then CWD may increase. We think that fire suppression has not been sufficiently successful to raise the amount of CWD to above the HRV (low confidence). This opinion is based partly on the considerable longevity of the trees that dominate the low-elevation landscapes (ponderosa pine, Douglas-fir, and limber pine) and the fact that pathogenic problems are still rather localized where these species occur.

5.2.4. Forest floor depth

Surface fires in low-elevation forests would undoubtedly burn much of the forest floor, and because they appear to be more frequent than stand-replacing crown fires, it is plausible that the forest floor (litter and duff) in such forests did not become as deep as in high-elevation forests during the reference period (1600-1850). Fire suppression probably is causing the development of deeper forest floors than normally would be expected, pushing this variable beyond its range of means during the HRV period (high confidence). A deeper forest floor in such forests probably contributes to the flammability of the forest, as does the increased tree density and more continuous fuel loads, which increase the potential for hotter surface fires that may kill more of the larger trees than previously.

5.2.5. Understory plant cover, species composition and diversity

Where trees and saplings have become more dense, tree canopy cover increases and there is a concomitant reduction in the amount of understory cover (Moir 1966, McPherson 1992). Also, shade-tolerant species become more abundant as canopy cover increases. In areas where tree density has increased or where harvesting has not thinned the trees, it seems probable that the percent cover of understory vegetation has been pushed below the HRV (low confidence). In some areas, exotic plants such as cheatgrass have invaded (such as near Woods Landing), but currently this is not thought to be a significant problem on the MBNF as a whole (Robert Mountain, personal communication). No data are available to suggest that plant species diversity is beyond the HRV, whether for trees or understory plants. Considering that the low elevation forests may have been quite dense at times, we think that species diversity probably is within the HRV (low confidence).

In summary, a combination of human influences appear to have caused several structural characteristics of ponderosa pine stands to exceed their HRV in other parts of the Rocky Mountains, and the same probably has occurred in the MBNF. Great attention has been given to this situation, often in the context of discussing "forest health." Low-elevation forests on the MBNF cover a small proportion of the terrain and provide a rather small amount of the wood that is harvested. However, they are important for wildlife habitat and recreation and they occupy a large proportion of the Sherman and Laramie Mountains units. Attention should be given to the potential adverse consequences of the unusual conditions now found in this forest type, where they occur, and to restoring the forest structure where appropriate (e.g., Romme et al. 2000).

5.3. Landscape structure at low-elevations

Very little information is available regarding the landscape structure of low-elevation forests and woodlands on the MBNF, both historically and at present. Romme et al. (2000) stress that information about landscape patterns in ponderosa pine forests is very limited throughout the Southern Rocky Mountains (which includes the MBNF).

Clearly, however, changes in landscape pattern are occurring in low-elevation forests dominated by ponderosa pine, especially in some parts of the Laramie Peak area. Due to fire suppression, logging, livestock grazing, and possibly climate change, not only are some of the forests becoming more dense, but ponderosa pine are invading some patches of grassland as well. Thus, forests and woodlands probably are occupying a larger area than previously in such habitats, while grasslands and savannas now occur on less of the landscape. Notably, research by Shinneman and Baker (1997) in the Black Hills suggests that this trend cannot be generalized wherever ponderosa pine occurs in the region. It should also be noted that ponderosa pine is commonly restricted to rocky outcrops on the fringes of the MBNF, where fires may be less frequent

(States 1968; Knight 1994a). When trees are clumped for edaphic reasons, they are not likely to grow into nearby grasslands.

Another factor contributing to landscape change is the harvesting of large trees, which promotes the growth of saplings. As a result, there probably are now more small trees than before in some areas, and, as small trees invade, tree cover probably has become greater and more uniform in parts of the Laramie Mountains unit. The net effect of fire suppression and harvesting has been a landscape that is less patchy and more homogeneous, with the result that snag density, CWD volume, and proportion of landscape in different cover types are beyond their range of means during the HRV period (high confidence).

This situation probably would pertain to Douglas-fir forests and woodlands as well, though this species is not common on the MBNF (see Figs. 19e and 23). Limber pine is not usually harvested for wood, and fire in limber pine communities is thought to be less frequent. Thus far, there is no evidence that limber pine distribution and density have changed because of human activity, though the invasion of the white pine blister rust is a concern in the Laramie Peak and Sherman Mountains units. The extensive outbreak of the mountain pine beetle on ponderosa pine from 1988 to 1994 could be within the HRV, as Shinneman and Baker (1997) found historical records for a beetle outbreak over much of the Black Hills in 1895 (with 90% of the trees killed in many areas).

5.4. Summary for low-elevation forests and woodlands

Overall, maintaining low-elevation forests within their HRV will require attention to the diversity of disturbance agents and processes with which these ecosystems appear to have developed. Under management strategies that include fire suppression, livestock grazing and the removal of large trees and snags, conditions will quickly depart from the HRV in some areas, if they have not already. On the other hand, strategies that recognize the importance of fire, insects, and other natural disturbances, and aim to maintain large, old trees and snags, will most likely allow low-elevation forests to be restored to their HRV.

Romme et al. (2000) suggest that dense stands of ponderosa pine are not necessarily bad, as they undoubtedly have always occurred on sites where, for whatever reason, surface fires may have been excluded for perhaps 40-50 years. They note, however, that such stands were most likely interspersed on the landscape with grasslands and savannas where surface fires occurred more frequently. The use of such fires as a management tool has become increasingly difficult as more people construct cabins and homes at low elevations. Notably, widespread fire suppression at high and low elevations has the potential of affecting a larger area than timber harvesting.

6. THE HRV OF ASPEN FORESTS AND WOODLANDS

Aspen usually is found in small, clonal groves throughout the MBNF, primarily at middle and low elevations (Fig. 30). An exception to this pattern is in the Sierra Madre unit, where aspen dominates 20% of the forested area and occurs in very large stands west of the Continental Divide. By comparison, aspen dominates 16% of the forests on the Sherman Mountains unit, 4% on the Snowy Range unit, and 1% on the Laramie Mountains unit (von Ahlefeldt and Speas 1996)—always in small groves. Usually aspen is found from 2130-2590 m (7,000 to 8,500 ft) in ravines or other concave surfaces where the soil is fine textured and moisture is available for a longer time during the summer. Notably, aspen is the only broadleaf, deciduous tree that occurs on the uplands of the MBNF. Though deciduous, aspen is evergreen in the sense that its bark has chlorophyll and is capable of contributing to photosynthesis in the spring before leaves develop and in the autumn after the leaves have fallen.

Aspen stands typically have relatively open canopies, allowing light to reach the forest floor. Lush herbaceous understories develop, primarily because of the light and the more abundant moisture and nutrient availability on sites where aspen becomes established. Alexander et al. (1986) described three aspen habitat types on the MBNF with different understory characteristics—the aspen/Fendler’s meadow rue (Thalictrum fendleri) and aspen/pine reedgrass (Calamagrostis rubescens) habitat types on moist sites with relatively deep soils, primarily in the Sierra Madre and Snowy Range units, and the aspen/elk sedge (Carex geyeri) habitat type on somewhat drier sites throughout the MBNF (though most common in the Sierra Madre unit).

The extensive stands of large aspen on the west slopes of the Sierra Madre occur where winter and summer precipitation is apparently higher because of the Arizona monsoon (Knight 1994a). Aspen dominates approximately 20,308 ha (50,140 acres) in the Sierra Madre, the largest area of contiguous aspen forest in Wyoming. The present stands (clones) appear to have originated following extensive fires in the late 1800s near Battle Mountain, though the forests may have existed before the fires as well. A large proportion of the aspen stands in all four units of the MBNF are approximately 100 years old (see 19c), because of extensive disturbances during the late 1800s and early 1900s. Alternatively, the abundance of 100-yr-old stands may be attributable to the fact that aspen trees (the shoots or ramets of the aspen plant) usually live only about 100 years and there may not have been a stand replacing disturbance in the last century.

Aspen reproduction typically is asexual, with new shoots produced from root sprouts—a process sometimes referred to as suckering (Barnes 1966, Bartos et al. 1991). This, combined with the way aspen is able to persist in the understory of some mature forests, explains why aspen tends to develop where it occurred previously. Sexual reproduction is quite rare, though seedlings do occur when severe disturbances are followed by the extended, moist conditions required for seedling establishment (McDonough 1985, Romme et al. 1997). Because of this, sexual reproduction is thought to be rare and episodic (DeByle and Winokur 1985, Romme et al. 1997). There is considerable genetic diversity between clones, with some clones adapted for higher elevations and some responding differently to weather conditions than others (Jelinkski and Cheliak 1992). For example, it is common to see two adjacent aspen stands (clones) in the fall, one with yellow leaves and the other with green leaves. The clones are thought to be very long-lived, perhaps thousands of years (Veblen and Lorenz 1991). However, the oldest trees (ramets) in climax aspen communities on the MBNF were about 180 years old in the mid-1980s (see Fig. 19c). Some aspen stands on the MBNF appear to be self-perpetuating (Alexander et al. 1986); others have an understory of conifer saplings—especially subalpine fir and lodgepole pine.

Aspen forests support a diversity of birds and small mammals, as good forage and ground cover is provided. Loose and Anderson (1995) found woodpecker nests more commonly in aspen forests than in other forest types, and Struempf (1999) found higher nest survival in aspen than in conifer forests. Large aspen >18 cm DBH (average 26.7 cm; range 18-45 cm) with heart rot or dead tops are frequently occupied by nesting woodpeckers (Loose and Anderson 1995). The largest aspen recorded by Severson (1963) on the MBNF was 66 cm DBH. The mean DBH of aspen in Loose and Anderson's study area in the Snowy Range was 11.8 cm. The plant species composition of most aspen forests is much different from coniferous forests, and consequently the animal species composition is different as well. While aspen does not occupy a large area on any of the units of the MBNF except the Sierra Madre, this species contributes significantly to the overall biological and aesthetic diversity of the landscape.

6.1. Disturbance history, human influences and succession

Aspen can be seral or climax depending on site conditions and historical factors (Mueggler 1985), but typically it is thought of as a pioneer species (Bartos and Campbell 1998). Usually a few aspen persist in the understory of coniferous forests and produce an abundance of new sprouts when the next stand-replacing disturbance occurs. Notably, while young aspen shoots within a clone share the same root system with the mature canopy dominants, few are able to survive in the understory.

Extensive fires during the settlement period appear to have created large stands of aspen, both in Colorado (Veblen and Lorenz 1986) and on the west slope of the Sierra Madre. It is unknown if these large stands developed largely from sprouts or seedlings. It is also unclear whether aspen or conifers were dominant prior to the last fires (Severson 1963). With attempts at fire suppression in recent decades, along with natural succession, conifers are now becoming co-dominants with aspen in some areas, causing some managers to recommend that the conifers be harvested or the whole stands burned to restore the aspen grove. In other areas, stands of aspen are not being invaded by conifers and appear to be relatively stable communities (Alexander et al. 1986, Mueggler 1985, Shepperd et al. 2001). Thus far there we have found no evidence that the amount of land area dominated by aspen on the MBNF is either lower or higher than the HRV (moderate confidence).

As with other forest types, various kinds of disturbances play a role in aspen dynamics, including wind-throw, snow breakage, insect epidemics, disease, fire, timber harvesting, and grazing and browsing by domestic and native ungulates. Aspen has relatively weak stems and brittle branches, and is therefore easily damaged by heavy snow accumulation, avalanches, and wind (Veblen and Lorenz 1991). Various insects defoliate aspen trees, but two of special concern are the forest tent caterpillar (Malacosoma disstria) and the large aspen tortrix (Choristoneura conflicta) (Allen and Harris 1999). Both can affect large areas over longer periods than most other insects. After the overwintering eggs hatch, the larvae of the tent caterpillar feed on the growing leaves. When population sizes are high, this caterpillar quickly defoliates entire trees and stands—usually above 2424 m (8,000 ft). The trees may produce new leaves, but infestation for several years in a row is lethal. When this happens, new aspen groves develop by sprouting or the conifer saplings in the understory grow into the canopy more rapidly. Epidemics occur every 10 to 20 years in Northern New Mexico and Southern Colorado, with a single epidemic lasting for up to 10 years or longer (Hinds 1976). Unfortunately, we have not found comparable information for the MBNF.

The tortrix similarly has larvae that feed at the time of leaf expansion, but this insect also feeds on buds and can reach epidemic population sizes for two to three years (Allen and Harris 1999). No epidemics of these insects have been reported for the MBNF.

In general, there is no basis to conclude that the characteristics of insect outbreaks in aspen stands are beyond their HRV (moderate confidence).

With regard to diseases, aspen are affected by a variety of fungi and viruses, including leaf blights, stem and root rots, and bark cankers (Allen and Harris 1999). The known diseases are native and tend to spread when the thin bark of aspen is damaged by animals, falling trees, humans, or other agents (Hinds 1976). Some diseases appear to increase with wounding associated with timber harvesting and campers (Hinds 1976, Walters et al. 1982), but conversion of older stands to younger stands may have reduced the occurrence of disease at a regional scale. There are numerous mechanisms whereby aspen diseases can be spread, both natural and anthropogenic, and consequently it is still impossible to determine if they are within the HRV.

Mean fire return intervals for aspen groves probably are essentially the same as for the other forest types with which they occur. Veblen et al. (1994) found fire return intervals in aspen forests of 160 and 240 years (mean = 202) in Colorado, which is similar to some nearby coniferous forests. While aspen is not considered to form a highly flammable forest, it often burns when the adjacent coniferous forests burn (DeByle et al. 1987, 1989). Lightning-caused ignitions in aspen stands are probably rare, but fire has definitely played an important role in the establishment of new aspen stands (Brown and DeByle 1989, Romme et al. 1997). If fires are of high intensity, some aspen root systems may be killed, thus favoring development of conifer stands (Parker and Parker 1983). However, the relatively mesophytic herbaceous understory in many aspen stands probably results more often in cooler fires than in adjacent stands of conifers. Such fires favor the development of large numbers of aspen root sprouts even though the aboveground part of the older trees (shoots) is killed (Veblen and Lorenz 1991).

As with coniferous forests, modern fire suppression combined with fine fuel reduction by livestock grazing probably has contributed to the lengthening of fire return intervals in some aspen forests. Moreover, fire suppression has almost certainly reduced the amount of aspen forest in some areas because of succession to conifers (Bartos and Campbell 1998, Bartos 2001). When large fires burn in the future, whether because of weather, fuel conditions, or changes in fire management policies, aspen regeneration and seedling establishment probably will occur (Romme et al. 1997). In the absence of overbrowsing of aspen suckers by elk and/or livestock, such regeneration would likely lead to increases in the amount of aspen forest on the landscape, even if only temporary (i.e., for at least 100 years in areas where conifers are likely to invade again through succession).

Fluctuations in aspen cover in some areas can be attributed to heavy browsing by large populations of ungulates, especially elk (DeByle 1979; Kay 1990, 1993; Kay and Wagner 1996; Baker et al. 1997; Hessl 2002) and secondarily domestic livestock. This effect has been observed primarily in the vicinity of national parks where ungulate populations have been quite high in recent years, such as in YNP and Rocky Mountain National Park. Some observations indicate that the effects of browsing and trampling on aspen regeneration are common in the MBNF, particularly in the Laramie Mountains unit (von Ahlefeldt and Speas 1996, page 6-71).

Very little harvesting of aspen has occurred in the MBNF, though there are discussions about creating young stands of aspen through clearcutting as well as with prescribed burns. Thus far, only about 283 ha (700 acres) of the aspen on the west slope of the Sierra Madre have been harvested (<2% of the aspen forest in that area).

In summary, because of very little timber harvesting, the variables pertinent to the structure of modern day aspen stands are probably within their HRV on the MBNF (moderate confidence). At the landscape scale, there may be a smaller area in aspen at the present time, due to succession to conifers, but this too has undoubtedly fluctuated during the last several centuries. Thus, it appears that management guidelines for aspen can be driven more by what is desired than by concerns about conditions that have exceeded the HRV. The primary human effects on aspen groves have been fire suppression on the landscape as a whole and the concentration of large numbers of ungulates on winter ranges in the foothills. While possibly different than during much of the reference period, when fires may have burned over larger areas than they do today, and when native ungulates may have been abundant at times, the disturbance history and structure of aspen stands probably is within the HRV (moderate confidence).

In fact, fire suppression may not have had much effect, considering that large fires appear to be climatically driven phenomena (Bessie and Johnson 1995, Weir et al. 1995, Kipfmueller and Baker 2000) which humans typically cannot control. Big game populations probably fluctuated as well (DeByle 1979), with high populations occurring after a series of mild winters, or when predation by wolves, mountain lions, and bear was low. Information pertinent to the debate about possible causes of changes in the abundance of aspen has been summarized in Shepperd et al. (2001) and Hessl (2002). It seems clear that a trend and its causes in one portion of the range of this widespread species are not necessarily the same in other areas. Much has been learned since the review of aspen ecology edited by DeByle and Winokur (1985), but much is still unknown about the long-term dynamics of this species in specific areas such as the MBNF.

7. THE HRV OF NON-FOREST VEGETATION

Though not as abundant as forests, different kinds of grasslands, shrublands and forblands are found throughout the MBNF (Table 3; distribution maps available from the MBNF RIS). Just as in forests, changes in structure and composition may occur through time as a result of both natural and anthropogenic factors. The non-forest vegetation types are treated in less detail than the forests, primarily because there has been less research done on them that enables an HRV interpretation.

Grasslands (including grass-dominated meadows) are found on the MBNF from the foothills to the upper limits of alpine tundra. Bluebunch wheatgrass (Elymus spicatum) is a characteristic species at low elevations with 25-50 cm of precipitation annually (Williams 1961, 1963). At cooler, more mesic high-elevation sites, Idaho fescue (Festuca idahoensis) becomes more common, often associated with shrubs such as threetip sagebrush (Artemisia tripartita). Wet meadows at high elevations have an abundance of alpine hairgrass (Deschampsia caespitosa).

In general, grasslands add variety to the landscape mosaic, occurring in places where the soil is too shallow, too dry, too fine textured, or too wet to allow tree seedling establishment (Knight 1994a, Doering and Reider 1992). The physical location of many mountain meadows and grasslands has likely been stable for centuries due to site-specific abiotic conditions (Doering and Reider 1992).

In some instances, however, changing climatic or edaphic conditions may allow tree seedling establishment on areas that were historically occupied by meadows. Some examples include the establishment of lodgepole pine in subalpine meadows in Yellowstone National Park (Jakubos and Romme 1993), and the growth of subalpine fir and Engelmann spruce near treeline in Rocky Mountain National Park (Hessl and Baker 1997). In both cases, climate changes after the end of the Little Ice Age (c. 1870) apparently created favorable conditions for tree seedling establishment. Of course, the converse of this also may be true where climatic conditions following forest disturbances lead to the development (at least temporarily) of a mountain meadow (Billings 1969). Therefore, while abiotic conditions can maintain relatively stable grasslands for long periods of time, the distribution of mountain meadows and grasslands undoubtedly changes as the climate changes.

On the MBNF, grasslands are scattered throughout all four units. In the Snowy Range and Sierra Madre they occur primarily as small patches interspersed among lodgepole pine and spruce/fir forests, with the exception of a few large parks (e.g., Cinnabar Park, Dry Park, and Headquarters Park). Alpine grasslands, a form of tundra, are found directly surrounding the highest peaks of the Snowy Range, blending in with patches of krummholz composed primarily of stunted, wind-swept Engelmann spruce, subalpine fir, and, less frequently, limber pine. In the Laramie Mountains, grasslands are interspersed with ponderosa pine, lodgepole pine, limber pine, Douglas-fir, aspen, and upland shrubs. Much of the Pole Mountain unit is often described as grassland, though the abundance of the short three-tip sagebrush suggests that this vegetation type should be classified as a shrubland.

Succession in mountain grasslands is difficult to evaluate, but Costello and Schwan (1946) concluded that an association of Idaho fescue and mountain muhly (Muhlenbergia montana) is a subclimax vegetation type where ponderosa pine occurs in the Rocky Mountains. Shifts in species composition can occur as a result of grazing by large mammals. The review by Johnson (1956) of the effects of grazing by big game and livestock on pine-bunchgrass rangelands in Central Colorado is pertinent to the MBNF.

Various kinds of shrublands are also common. Dominant species listed in the RIS database include serviceberry (Amelanchier alnifolia), sagebrush (mostly mountain big sagebrush, Artemisia tridentata var. vaseyana), threetip sagebrush (Artemisia tripartita), fringed sagewort (Artemisia frigida), mountain mahogany (Cercocarpus montanus), antelope bitterbrush (Purshia tridentata), shrubby cinquefoil (Pentaphylloides floribunda), greasewood (Sarcobatus vermiculatus), and snowberrry (Symphoricarpos albus, occidentalis, and oreophilus). Additional species include chokecherry (Prunus virginiana), snowbrush ceanothus (Ceanothus velutinus) and wild rose (Rosa spp.). In general, all of the shrubs are characteristic of the mixed foothill shrubland cover-type (Knight 1994a). Their distribution is controlled largely by climate, elevation, and topographic factors.

Mountain big sagebrush, for example, often occurs where moisture is available but the soils are well drained and temperatures are not too cold (Burke et al. 1989). Mountain mahogany is found primarily on rocky, sedimentary substrates in the Sherman Mountains unit (Brooks 1962, Knight 1994a, von Ahlefeldt and Speas 1996). Bitterbrush and snowberry often are mixed with mountain big sagebrush in the foothills, especially at lower elevations where snow accumulates, such as in ravines. At higher elevations and in association with wet meadows, shrublands dominated by mountain silver sagebrush and shrubby potentilla are common. In the eastern portions of the Laramie Mountains unit, at lower elevations, skunkbush sumac (Rhus trilobata) sometimes occurs with bitterbrush. In riparian zones, shrublands dominated by willow (Salix spp.) are common throughout the MBNF.

Another type of shrubland (or woodland) on the MBNF is dominated by Gambel oak (Quercus gambelii). Perhaps because its seedlings are apparently not able to tolerate spring frosts and low summer precipitation (Neilson and Wullstein 1983), the distribution of Gambel oak on the MBNF is limited to relatively low elevations of 1,675 to 1,980 m (5,500-6,500 ft), on the southwest side of the Sierra Madre near the Colorado state line (Knight 1994a). Gambel oak survival in this area probably is the result of increased summer precipitation due to the influence of the Arizona Monsoon combined with relatively warmer temperatures below 1,980 m (6,500 ft). Further south, Gambel oak commonly occurs with ponderosa pine. Such woodlands can become quite flammable, but the oak sprouts vigorously and may increase in density after fires (Harper et al. 1985). With fire suppression, the oak woodlands may be invaded by Rocky Mountain juniper.

Other types of shrubland are found only at high elevations in the MBNF. Where the subalpine and alpine zones come together, subalpine fir, Engelmann spruce and sometimes limber pine form wind-swept, shrubby patches referred to as krummholz, which is classified as a shrubland in the USFS-RIS database. In the alpine zone, shrublands of alpine dryad and willow occur in areas that are relatively moist or protected from the wind.

The alpine area surrounding Medicine Bow Peak provides an important element of biological diversity. Numerous studies have been conducted on the geology, soils, vegetation, and grazing of this relatively small alpine area (e.g., Ray 1940; Mears 1953, 1962; Bliss 1956; Billings and Bliss 1959; Billings and Mooney 1959; Johnson 1962, Billings 1969, Thilenius 1975), many of which are reviewed in Knight (1994a) and von Ahlefeldt and Speas (1996). In general, the alpine vegetation of the Medicine Bow Peak area is dominated by grasses, shrubs, and forbs that are able to withstand the severe environment characterized by high winds, low humidity, cold soil temperatures, high ultraviolet radiation, short growing season, low soil moisture, and great daily temperature fluctuations (Bliss 1956, Knight 1994a).

7.1. The effects of disturbances on non-forest vegetation

As with forest vegetation, the composition and structure of non-forest cover types is affected by both natural and human-caused disturbances. Fire and herbivory have been the primary historical disturbance agents in non-forest vegetation, but the characteristics of both have changed in the past century, primarily due to the introduction of fire suppression and grazing by domestic livestock.

Fire suppression probably has been in place long enough to cause comparatively high levels of shrub cover and the spread of trees into some shrublands and grasslands. Indeed, the USFS now uses prescribed burning to reduce the cover of woody plants (Cook et al. 1994), which stimulates the growth of herbaceous plants and slows the invasion of other woody species, such as Rocky Mountain juniper, aspen, and various species of conifers. Long periods without fires could have occurred during the reference period as well, at least in some areas. If true, shrub and tree cover are still within the HRV (moderate confidence).

Where big sagebrush is a dominant species, vegetation changes can be quite dramatic after fire because this species is not capable of sprouting. When a fire occurs, grasses and forbs become the dominants for periods of several decades. Within approximately 10 years, however, big sagebrush usually will reinvade as new seedlings become established. Some grasses, such as bluebunch wheatgrass, may also decline in abundance following hot fires (or heavy spring grazing), but they, too, usually recover. In contrast, the other shrub species often resprout after a fire, such as Gambel oak, serviceberry, bitterbrush, and snowberry (Cook et al. 1994). Consequently, their cover is rapidly restored, and sometimes their density increases to above pre-fire levels (Harrington 1985, 1987). Sprouting is an important adaptation for surviving fires, but some species are adapted in other ways as well. For example, the heat of fire facilitates the breaking of snowbush ceanothus seed dormancy (Zavitkovski and Newton 1968).

As shrub cover increases with time since fire, the flammability of the vegetation also increases due to increased amounts of small woody fuels. Prescribed fires can help prevent wildfires, but the presence of ranches, summer homes, and resorts in aesthetically pleasing foothill environments often restrict the implementation of this tool.

Bison and other herbivores probably grazed montane grasslands prior to the arrival of domestic livestock. If true, moderate grazing probably would not have driven variables beyond their HRV. However, sheep grazing at high elevations in the early 1900s appears to have exceeded the intensity of grazing by native herbivores. A long-term manager of the Sierra Madre unit, James Blackhall, wrote in 1916: “The grazing industry of this section is nearly extinct, brought on by overstocking, in fact with the exception of that portion which is protected by the long period that the snow lies on the ground, the short summer season and the early advent of winter, it may be said that the country is destitute of all forage necessary to support animal life . . .” (Bruce 1959, p. 51). Other parts of the MBNF probably experienced heavy livestock grazing as well during this time, pushing at that time various ecosystem variables beyond their range of means during the HRV reference period (high confidence).

Such heavy grazing intensities no longer occur on the MBNF (see Section 3.3) and there is evidence that the vegetation is recovering. Still, livestock grazing was so widespread for so long that the effects are difficult to identify, even for trained observers (Thilenius 1975, Turner and Paulsen 1976, Fleischner 1994). Some plant species now may be quite rare because of livestock grazing, and some soil characteristics may not have been restored. Domestic livestock are thought to have different grazing and browsing habits than native ungulates, which could have accentuated the adverse effects of very high numbers. Our observations suggest that rangelands appear to be in fair to good condition in most areas, both in the upland and in riparian zones, but further research is necessary to know if the ecosystems are again within the HRV.

Other disturbances that affect the dynamics of non-forest vegetation are pocket gophers and the invasion of exotic plants. The burrowing of pocket gophers can be conspicuous, where the animals bring large amounts of soil to the surface. Some of the displaced soil is in elongated mounds (middens), which are created during the winter as soil is moved into the animals' snow tunnels. This burrowing greatly softens and aerates the soil profile. We have no evidence to suggest that the densities of pocket gophers are within or beyond their HRV in different localities.

In contrast, exotic plants now are common in the meadows and shrublands of the MBNF, and some stands of aspen. Disturbances that accompany forest restoration, timber harvesting, livestock grazing, high intensity recreation, and road construction may facilitate the establishment and spread of exotic plants. Rapid increases in such plants can cause a decline in the abundance of some native plants and can cause changes to other ecosystem processes, such as the spread of fires. Exotic plants are still a minor problem in the post-harvest forests of the MBNF (Selmants 2000), but dramatic changes in non-forest vegetation could occur if exotic plants such as cheatgrass become more abundant (Keane et al. 2002). The introduced bluegrasses may have less obvious effects on ecosystem variables, but in general their effects are still poorly known. Recent research on Rocky Mountain grasslands suggests that exotic plants are most likely to invade ecosystems where native plant diversity is high (Stohlgren et al. 1999a), because of the favorable soil conditions found in such places, and that grazing probably has little effect on the accelerated spread of most exotic plant species across the landscape (Stohlgren et al. 1999b).

In summary, intense livestock grazing in the early part of the 1900s, and fire suppression and the introduction of exotic plants in the mid-1900s, have caused changes over the years to non-forest ecosystems in some parts of the MBNF, probably pushing some variables beyond their HRV. Livestock grazing has been greatly reduced in recent decades, and many of the rangelands probably are in better condition than before. However, insufficient research has been done to determine if some present-day ecosystem characteristics can be attributed to heavy livestock grazing in the past. Fire suppression during the last 50 years or more has caused changes as well, but there may have been comparable periods without fires during the 1600s and 1700s. The adverse effects of exotic plants could become more severe in the future. Compared to grazing intensity and fire suppression, the spread of exotic plants is more difficult to control. However, the adverse effects of exotic plants are more subtle.

8. SUMMARY OF PROBABLE HRV DEVIATIONS

Based on the information that we reviewed, and our understanding of the ecology of the MBNF, it seems clear that some ecosystem variables could be outside their HRV or trending in that direction in some areas where timber harvesting, livestock grazing, fire suppression, and the introduction of exotic plants have occurred. The removal of large wood is a type of disturbance that is different from any other with which the forest ecosystems have developed, with the result that both stand structure and landscape structure have been affected in some areas. Largely as a consequence of timber harvesting and fire suppression, ecosystem development is now proceeding in such areas in ways that are different than prior to the advent of current forest management practices. Does this matter? An answer is beyond the scope of this report. However, deviations from the HRV, where they occur, should be evaluated to determine if they could lead to undesirable consequences.

As our report shows, disturbances to the forests of the MBNF are not new. Indeed, the forest species have evolved with regular disturbances and the historical record indicates that there was a significant wave of human-caused disturbance in the late 1800s and early 1900s. Current management practices often are occurring on forests that have already been subjected to at least one episode of timber harvesting and numerous episodes of burning. For this and other reasons that we have reviewed, there is great spatial and temporal variation in most ecosystem properties. It is reasonable to argue that the species that exist on the MBNF in the year 2000 are those that survived rather intensive harvesting and burning caused by European-Americans during the late 1800s and early to mid-1900s. A century, however, is a short time in the development of forest ecosystems, and some of the effects of humans in the early 1900s could be augmented by the additional road construction and harvesting that have occurred since that time.

Based on our understanding of MBNF ecosystems, we have attempted to draw conclusions about whether or not certain ecosystem variables are now beyond their HRV during the period 1600-1850, or trending in that direction (Table 7). To summarize, we believe there is evidence to suggest that the following forest variables are within the HRV at high-elevations:

Extensive, stand-replacing fires
Insect population sizes
Abundance of diseases (including dwarf mistletoe)
Density of trees in all size classes (though the mean density may have been lowered in some areas)
Tree and understory plant cover and diversity (number of species)
Root/shoot ratios
Forest floor depth
Proportion of the landscape in different cover types
Ratio of forest to non-forest land

Similarly, the following forest variables appear to be within the HRV in aspen forests and low-elevation landscapes:

Aspen stand abundance and stand structure
Abundance of native insects and diseases
Understory plant species diversity (number of species)

Inadequate information exists for drawing conclusions for non-forest vegetation types.

In contrast, we believe other ecosystem variables are either beyond their HRV now or could be in the future if current management practices continue, as summarized below and in Table 7.

Within stands at high elevations affected by timber harvest

1. Tree cover probably is lower and the number and size of canopy gaps probably is higher in harvested stands than the range of means for these variables during the HRV period in unmanaged forests of comparable age and site conditions (moderate confidence), due to selective and shelterwood cuts, and fewer standing-dead trees,
2. Where timber harvesting has occurred on the MBNF, snag density and the amount of coarse woody debris is lower than the range of means for these variables during the HRV reference period for unmanaged stands of comparable age and site conditions (high confidence). Whole-tree yarding can accentuate this variation by concentrating coarse woody debris and slash at landings in a way that has not occurred previously.
3. Due to some methods of slash treatment, the intensity of soil scarification is beyond the HRV in some areas (high confidence). Burning can remove the forest floor, but the depth of soil disturbance after fire typically is less than following mechanical harvesting.
4. Age- and size-structure of managed stands within ravines or on north or leeward slopes is skewed toward smaller and younger trees than would have occurred during the HRV period, with less old forest, because of harvesting where fires would have been less likely to burn (moderate confidence).

5. Dwarf mistletoe abundance could be above its HRV in stands subjected to fire suppresion and partial timber harvests that did not remove infected trees (moderate confidence).

Across the landscape at high elevations

6. The estimated rotation times for timber harvest of forests judged suitable for harvesting are generally shorter, on average, than the estimated site-specific mean intervals between natural disturbances in high-elevation forests (moderate confidence).
7. Old-growth forest is less common, especially in areas that have had long fire-free intervals, such as in valley bottoms and on some north or leeward slopes. Thus, ecosystem characteristics such as snags and CWD that depend on long periods of forest development may be less common across the landscape than during the reference period (high confidence).
8. The rate of patch formation and the size of disturbances across high elevation landscapes exceed their HRVs (moderate confidence). Timber harvest produces patches more frequently than natural disturbances, and mid-sized patches (i.e., 1-20 ha) probably are more common in the landscape now than during the reference period. Natural fires produce many small patches at short intervals and a few very large patches at long intervals at high elevations. Fire frequency, size, and extent have been suppressed, but probably not to the point of exceeding the HRV (moderate confidence)
9. Due to roads and timber harvesting, the amount of edge is now higher and the land area of interior forest is lower, on average, than prior to the 1900s (high confidence). Correlated with this is a decline in the average patch size, the development of a more simple patch shape (with more abrupt edges), less connectivity among patches of old forest, and a higher, more uniform level of landscape patchiness than would have occurred during the reference period.

10. The proportion of the forested land with high tree density is somewhat lower because of partial thinning designed to maximize rates of tree growth (moderate confidence)

Within stands at low elevations affected by harvest, fire suppression and livestock grazing

11. Fire suppression, livestock grazing, timber harvesting, and possibly climate change have led to the development of low-elevation forest stands with high densities of young trees, greater canopy cover, and less understory than usually occurred on many sites before 1850 (moderate confidence). Understory composition has also changed (low confidence) and forest floor depth has increased (high confidence).
12. By removing large trees and downed wood from the system, wood harvesting has lowered the abundance of snags and coarse woody debris in managed stands of low-elevation forests (high confidence). Harvesting in recent decades, where it has occurred, also has led to an increase in sapling density and a reduction in the average diameter of trees, thus creating younger, more uniform-sized stands than existed before harvest (moderate confidence). Much of the harvesting affecting low-elevation forests on the MBNF occurred in the late 1800s and early 1900s.
13. Mean fire return intervals have become longer than the HRV at low elevations on drier sites, but fire intensity is probably higher due to the amount and continuity of fuels that could lead to more stand-replacing fires in the future (moderate confidence).
14. Forest floor depth may be deeper than the HRV at low elevations where fire suppression has been successful and prescribed burning has not been done (moderate confidence)
15. White pine blister rust outbreaks exceed the HRV in affected stands, as this disease is not native (high confidence).

Across the landscape at low elevations

16. Fire suppression, livestock grazing, and the logging of older, more fire resistant trees, where this has occurred at low elevations, probably has increased the amount and continuity of fuels to a point that is above the HRV, leading to potentially more stand-replacing fires than occurred prior to 1850 (moderate confidence). Climate warming may contribute to this trend.
17. Fire suppression has reduced the level of interspersion of tree stands with grasslands (low confidence), and the average tree/sapling density has increased above the HRV (moderate confidence).

Aspen forests

18. In some areas, grazing by livestock and native ungulates has reduced aspen densities and increased the abundance of non-native species in the understory of aspen stands, to the point where such variables are beyond their HRVs (moderate confidence). Some aspen stands now have an understory of conifers, but this succession is natural and probably occurred during the reference period as well after long periods without fire.

Non-forest vegetation

19. Some meadows, grasslands, and shrublands are probably grazed more, or have been in the past, than during the HRV period, though reductions in livestock numbers since the early 1900s have been dramatic and rangeland condition is improving. Where heavy grazing had occurred, some species probably declined in abundance, some probably became more abundant, and soil erosion probably occurred more rapidly (high confidence). Deviations from the HRV also were caused by the invasion of exotic plants that displace native species and cause changes in other ecosystem properties, but thus far their effects are not well known on the MBNF.

Frequently there has been a lack of pertinent quantitative data for the ecosystem variables we addressed in this report. Such variables can be identified as priorities for future research. Research will also be required if decisions about the severity of a problem cannot be made without more information or if ways of restoring the HRV cannot be identified.

Where deviations from the HRV do occur, various approaches can be used to work toward the goal of bringing variables within the HRV, if that is desired. In some cases, this may take many years to achieve. The challenge of bringing landscape variables within the HRV is likely to be greater on lands classified as suitable for timber harvesting (about 55% of the MBNF at the present time). Such lands are typically the most productive and accessible, and therefore they have the highest potential for both timber production and for supporting the elements of landscape structure that are in decline, namely, large blocks of contiguous forest with relatively large trees.

As noted, it is beyond the scope of this report to evaluate the possible ecological consequences of exceeding the HRV. However, HRV analyses can guide discussions in the future as managers continue to develop ways of using Rocky Mountain forests in a sustainable way. Specific site conditions and management practices must be considered. Much has been learned about forest ecosystems, but the kinds of influences to which national forests are now subjected are different and occurring more rapidly than ever before—and in a cumulative way. Human experience with modern forest management has been too short to speak with confidence about sustainability, but HRV analyses can help achieve that goal. Our report, which focuses primarily on upland vegetation, should be supplemented with HRV reports on riparian and other wetland ecosystems, and the population dynamics of plant and animal species thought to be potentially threatened by management activities.

The kinds of deviations that we have identified are not unique to the MBNF. However, the MBNF and the Black Hills National Forest are the major sources of wood in Wyoming. The Black Hills have high rates of wood production because of the moderate climate and long growing season at low elevations. The MBNF is higher in elevation and less favorable for tree growth, but its terrain is comparatively flat and easily accessible, compared to, for example, the Shoshone National Forest. Also, much of the land area is forested, in contrast to the Big Horn National Forest, and only a small area is roadless, in contrast to the Shoshone National Forest and Bridger-Teton National Forest. Another contributing factor for the comparatively high rate of wood harvesting in the MBNF may be the close proximity of major markets for wood on the Colorado Front Range. Notably, those markets also have created demands for aesthetic and recreational amenities. No longer is it an easy decision to convert old forests with slow tree growth to young forests with rapid wood production. Also, because of the construction of second homes adjacent to national forests, no longer is it an easy decision to use surface fires to simulate natural processes, with the goal of maintaining the HRV for various attributes of stand structure.

Timber harvesting is sometimes promoted as a way of solving problems thought to be associated with fire suppression and the development of what appear to be unhealthy forests. Indeed, human values 100 years ago set in motion changes that have pushed some ecosystem variables outside their HRV. The high density of ponderosa pine in some areas is an example. However, sick and dying trees surely have always been a part of the MBNF—sometimes over large areas. Such trees create habitat for organisms that cannot survive on healthy trees. The biological diversity of the forest, which assures the long-term survival of forest ecosystems without human intervention, is higher because of periodic waves of mortality. Disturbances caused by European-Americans, however, can be different than those with which the native species have evolved in recent centuries. A consideration of the HRV of various variables can be helpful in planning for the future.

For some observers of management practices, forests labeled as unhealthy should be treated through harvesting the diseased and slow growing trees, if not all of the trees. That may be desirable in some situations, but such judgements should be preceded by analyses to determine whether or not fire suppression really has been effective and whether or not the "treatment" creates more problems than it solves. A stand of healthy trees surely is the goal of a tree farm on private land, but widespread stands of fast growing trees over much of a National Forest, where the pathogens are native, would not be within the HRV. Similarly, to harvest a forest to the point where it could not burn would exceed the HRV for some ecosystem variables. The continuing challenge of forest management is to determine if there are options for reducing the undesirable effects that can be associated with extracting wood, suppressing fires, and raising livestock while providing the other resources desired by modern society.

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Table. 1. Chronology of major geologic and climatic changes in Wyoming. Mybp = million years before present; ybp = years before present. From literature reviewed by Knight (1994a) and von Ahlefeldt and Speas (1996).

TIME	CLIMATE/GEOLOGY	VEGETATION
350 mybp	Tropical climate; submerged under saltwater	Tropical
300 mybp	Swamps, lagoons, tidal flats	Clubmosses, horsetails, and ferns in swamps; pine, spruce and fir on upland
100 mybp	Ancestral Rocky Mountains leveled, still near equator	
245-66 mybp	Overthrust belt formed (~66 mybp)	Magnolia, palm, fig, breadfruit, sassafras, cinnamon, sweetgum, and willow in wetlands; conifers, ginkgo, and cycads on uplands
66-50 mybp	Mountains of MBNF uplifted during Laramide orogeny; continued drifting apart of continents; climate still tropical, snow rare, much wetter than today	Flowering plants more common in upland forests; species include alder, beech, birch, black locust, chestnut, cottonwood, cyprus, dogwood, elm, fir, ginkgo, hickory, maple, oak, pine, redwood, spruce, sweetgum, sycamore, walnut, and willow; palm and breadfruit also present
50-36 mybp	Tremendous erosion, volcanism, and basin filling	
10 mybp	Regional uplifting creating present mountain landscape	
7-5 mybp	Increased aridity, creation of rainshadows	Demise of some forests, giving rise to shrublands and grasslands; drought tolerant species persisted or immigrated; numbers of xerophytic plants increased in grasslands; fires became more frequent, contributing to the demise of trees and the spread of grassland; woodlands restricted to ravines, valley bottoms, and ridges
660,000 ybp	Volcanism (creating Yellowstone plateau); climate cooling, heavy snows, further cooling and formation of glaciers	
2 mybp - 10,000 ybp	Six glacial advances and retreats; flooding, creation of outwash plains, deposition of loess, and soil development	Coniferous forests, shrublands, and grasslands (many broad-leaved trees now regionally extinct)
127,000 ybp	Interglacial period	Douglas-fir and limber pine common in Yellowstone
15,000 ybp	Glacial advance; temperatures 10-13˚C colder than today; permafrost	Trees in Yellowstone confined to a narrow elevational band; lowlands tundra-like
11,500 ybp	Mean temperatures 5-6˚C cooler than today; retreat of glacial ice	Upper treeline about 600 m lower than today in Yellowstone; gradually colonized by Engelmann spruce, and later by subalpine fir and whitebark pine in some areas (11,000-9,500 ybp)
9,500 ybp	Continued warming	Establishment of lodgepole pine at higher elevations and Douglas-fir in foothills in Yellowstone
9,000-7,000 ybp		Upper treeline at its lowest in Colorado Front Range
7,000-4,000 ybp	Altithermal period, comparatively warm and dry conditions	Expansion of sagebrush, greasewood, juniper and grasses; grassland found up to 2,400 m on some north slopes in MBNF; spruce and fir retreated to higher elevations; elevational range of spruce and fir reduced
4,000 ybp - present	Neoglacial period of gradual cooling	Expansion of forests to previous elevational ranges
1350 – 1850 AD	Characterized by extended and extreme wet periods and droughts (Gray et al., in preparation)	
1850 AD - present	Slightly warmer than in previous centuries (Fastie et al., in preparation), and with shorter, less extreme wet periods and droughts (Gray et al., in preparation)	Increased tree recruitment near upper treeline (Hessl and Baker 1997), including subalpine meadows (Jakubos and Romme 1993)

Table 2. Upland vegetation types on the MBNF, classified at four levels of detail. ABLA = *Abies lasiocarpa* (subalpine fir), JUSC = *Juniperus scopulorum* (Rocky Mountain juniper), PICO = *Pinus contorta* (lodgepole pine), PIEN = *Picea engelmanii* (Engelman spruce), PIFL = *Pinus flexilis* (limber pine), PIPO = *Pinus ponderosa* (ponderosa pine), POTR = *Populus tremuloides* (aspen), PSME = *Pseudotsuga menziesii* (Douglar-fir), QUGA = *Quercus gambelii* (Gambel oak).

This report	**MBNF RIS**	**Alexander et al. 1986**	**von Ahlefeldt and Speas 1996**
High elevation forests	Engelman spruce-subalpine fir	3 ABLA habitat types	10 ABLA community types
			6 ABLA-PIEN community types
	Lodgepole pine		3 PIEN community types
	Limber pine	5 PICO habitat types	13 PICO community types
		2 PIFL habitat types	5 PIFL community types
Low elevation forests and woodlands	Ponderosa pine	3 PIPO habitat types	14 PIPO community types
	Limber pine	(see above)	(see above)
	Douglas-fir and mixed conifer	PSME series	9 PSME community types
	Rocky Mountain juniper		
	Pinyon – juniper [1]	JUSC series	4 JUSC community types
Aspen forest	Aspen – birch [1]	3 POTR habitat types	11 POTR community types
			7 POTR-ABLA community types
			1 POTR-PICO community type
			2 POTR-PSME community types
Non-forest vegetation (grasslands, shrublands and forblands)	Grasslands: wheatgrass, mountain muhly, needle-and-thread, tufted hairgrass, bluegrass		1 tufted hairgrass type 3 danthonia types 1 idaho fescue type 1 mountain muhly type 2 needle-and-thread types 2 western wheatgrass types 1 bluebunch wheatgrass type 4 sedgeland types 2 kobresia types[1]
	Shrublands: serviceberry, sagebrush, mountain mahogany, bitterbrush, alpine dryad, shrubby cinquefoil, greasewood, snowberry, thinleaf alder, Gambel oak, bur oak, krummholz;	QUGA series	4 serviceberry types 10 sagebrush types 5 mountain mahogany types 1 ocean spray type 4 antelope bitterbrush types 1 skunkbush sumac type 1 western snowberry type
	Forblands		3 QUGA community types
			2 mountain dryad types 3 alpine avens types 2 sibbaldia types

[1] Pinyon pine, kobresia, and birch trees (*Betula* spp.) and are not known to occur on the MBNF, though birch shrubs are common in some areas.

Table. 3. Acres and percent of total (in parentheses) of different land cover types on the four units of the Medicine Bow National Forest. To convert acres to hectares, multiply by 0.405. Source: MBNF-RIS

	Sierra Madre	**Snowy Range**	**Laramie Mtns**	**Sherman Mtns**
Forested land				
Aspen	59,800 (17%)	20,795 (3.8%)	2538 (1.4%)	759 (7.4%)
Cottonwood	310 (0.09)	10 (<0.01)	113 (0.06)	0
Douglas-fir	2374 (0.7	6105 (1.1)	1604 (0.9)	165 (1.6)
Limber pine	218 (0.06)	1583 (0.3)	9874 (5.4)	179 (1.8)
Juniper	178 (0.05)		70 (0.04)	
Ponderosa pine	28 (0.01)	8616 (1.6)	86,167 (47)	1032 (10)
Lodgepole pine	135,181 (39)	280,141 (50)	58,318 (32)	1682 (16.5)
Spruce/fir	73,387 (21)	117,927 (21)	2478 (1.4)	18 (0.2)
TOTAL	271,476 (78)	435,178 (79)	161,162 (89)	3835 (37.6)
Shrublands				
Gambel oak	1530 (0.4)	0	0	0
Sagebrush	44595 (13)	40,989 (7.4)	8280 (4.6)	1372 (13.4)
Mountain mahogany	4176 (1.2)	1979 (0.4)	183 (0.1)	
Riparian	1986 (0.6)	13,584 (2.5)	68 (0.03	213 (2.0)
Other shrublands	11,397 (3.3)	177 (<0.01)	3947 (2.2)	
TOTAL	63,684 (18.4)	56,729 (10.3)	12,478 (6.9)	1585 (15.5)
Grasslands/meadows	9005 (2.6)	42,736 (7.7)	5676 (3.1)	4758 (47)
Alpine/krummholz/barren	2108 (0.6)	8601 (1.6)	2635 (1.4)	27 (0.3)
Lakes, reservoirs, streams	643 (0.18)	1401 (0.3)	5 (<0.01)	2 (0.02)
TOTAL AREA	346,916	553,008	181,951	10,205

Table 4. Examples of successional stage characterizations for lodgepole pine and spruce-fir forests. See Table 2 for clarification of tree species abbreviations.

	SUCCESSIONALSTAGE	YEARS AFTER STAND-REPLACING DISTURBANCE	CHARACTERISTICS
Romme and Despain 1989; Despain and Romme 1991	LP0	0 to 30-50	Vegetative sprouting of herbaceous plants; PICO seedling establishment; large, dead tree boles that do not ignite easily, but sapling canopy near ground can carry fire during drought years
	LP1	30-50 to 150-200	Canopy closure; stands can be relatively dense with little groundlayer vegetation; period of tree growth, competition, and thinning; fuels are discontinuous from forest floor to canopy; does not burn readily except during periods with severe drought and wind
	LP2	150-200 to 200-400	Tree growth declines; some tree mortality creates canopy gaps; groudlayer more dense and more diverse; mixed understory of PICO, ABLA, and PIEN common; dead, woody fuels increase but still somewhat resistant to fire; fires are less intense in absence of wind and drought
	LP3	after 200-400	Canopy mortality increases; understory trees reach 2-3 m tall and penetrate canopy gaps; fuels and organic materials accumulate and burn easily and intensively during drought
Aplet et al. 1988	Colonization phase	0 to 100-200	Initial colonization by ABLA and PIEN simultaneously (and potentially PICO at some sites, although not in the study by Aplet et al.); period of initial tree growth and canopy closure
	Spruce exclusion phase	100-200 to 200-300 (lasting for ~100 years after initial colonization)	Dense canopy; PIEN unable to reproduce in understory
	Spruce reinitiation phase	200-300 to 300-400 (lasting for ~100 years after spruce exclusion)	Dominant ABLA and PIEN (and potentially PICO) begin to die, creating canopy gaps; PIEN (and possibly PICO) is able to reproduce in understory again
	Second generation spruce-fir forest	after 300-400	Uneven-aged canopy; cohort of spruce established during reinitiation phase reaches canopy; high fuel accumulation; very flammable during suitable summer weather conditions
USFS (1985)	Grass/forb/seedling	lodgepole (spruce-fir) 0 to 20 (0 to 25)	Colonization; size classes of trees is <2.5 cm DBH for PICO and ABLA-PIEN
	Sapling	21 to 40 (26 to 50)	Most trees 2.5 – 12.5 cm DBH
	Pole	41 to 80 (51 to 80)	Most trees 12.5 – 23 cm DBH
	Mature	81 to 140 (81 to 140)	Most trees >23 cm DBH
	Overmature	140+ (140+)	Most trees are larger than 33 cm

Table 5. Examples of ranges for mean fire intervals in high-elevation forests, as estimated by various investigators. The values are the estimated average time between a series of successive fire events in an individual stand (point-scale).

Author	**Forest type / location**	**Fire return interval estimate (years)**	**Type of fire event**	**Basis for estimates**
Billings (1969)	Spruce-fir / MBNF	up to 600-700	Stand-replacing	Stand age structure
Arno (1980)	Subalpine forest / N. Rockies	150-350	Stand-replacing	Fire scarred trees and stand age structure
Romme (1980a, 1982)	Lodgepole, fir, spruce / YNP	300+	Stand-replacing	Stand age structure and fuel accumulation
Romme and Knight (1981)	Spruce-fir / MBNF	300-500+	Stand-replacing	Stand age structure and fuel accumulation
Romme and Despain (1989)	Lodgepole, fir, spruce / YNP	200-400	Stand-replacing	Stand age structure and fuel accumulation
Hawkes (1980)	Lodgepole, fir, spruce, larch / Alberta	90-304 [1/]	all	Fire scarred trees
Kipfmueller (1997)	Lodgepole, fir, spruce / MBNF	39-149 [2/]	all	Fire scarred trees

[1/] This range is based on the means calculated for forests at different elevations and topographic positions, including: 1) lower subalpine/lower elevation (90-101 years); 2) upper subalpine/upper elevation (153-304 years); 3) north aspects (187 years); and 4) south, east and west aspects (93-104 years).

[2/] This range is based on four different methods for calculating point-scale MFI: 1) the interval between stand origin date and first scar (82 years); 2) the interval between scars on the same tree (39 years); 3) the interval between the last scar and the present (149 years); and 4) the overall average of all intervals (129 years).

Table 6. Examples of fire return intervals reported for stands (point-scale) of ponderosa pine and Douglas-fir in the Rocky Mountains. Baker and Ehle (2001) concluded that differences in such estimates are not significantly different.

Author	**Forest type / location**	**Fire return interval estimate (years)**	**Type of fire event**	**Basis for estimates**
Brown et al. (2000)	Ponderosa pine in Laramie Mtns unit of MBNF (Ashenfelder Basin)	Pre-settlement median intervals = 26 to 33.5; Range = 8 to 82	Surface	Fire scarred trees
Fisher et al. 1987	Ponderosa pine in Devils Tower Nat'l Monument, Wyoming	1600-1770: 27 1770-1900: 14 Pre-settlement: 19 Since 1900: 42	Surface	Fire scarred trees
Houston 1973	Douglas-fir in Yellowstone Nat'l Park, Wyoming	Pre-settlement: 20-25	Surface	Fire scarred trees
Laven et al. 1980	Ponderosa pine in Roosevelt Nat'l Forest, Colorado	3-161 (mean = 45.8)	Surface	Fire scarred trees
Goldblum and Veblen 1992	Ponderosa pine/Douglas-fir near Boulder, Colorado	3-49 Means for pre-1859: 32 1859-1920: 8 After 1920: 28	Surface	Fire scarred trees
Brown 1994; Brown and Sieg 1996	Ponderosa pine in Jewel Cave Nat'l Monument, South Dakota	9-79	Surface	Fire scarred trees
Swetnam and Baisan 1996	Ponderosa pine in Northern Arizona	Pre-settlement: 2-12	Surface	Fire scarred
McCune 1983	Douglas-fir in Montana	Pre-settlement: 60		
Arno and Gruell 1986	Douglas-fir in Montana	Pre-settlement: 35-40		
Shinneman and Baker 1997	Ponderosa pine in Black Hills Nat'l Forest, Wyoming and South Dakota	Pre-settlement: 250-300+	Crown	Historical observations; stand ages

Table 7. Summary of variables describing upland vegetation characteristics on the Medicine Bow National Forest (MBNF) relative to their estimated historic range of variability (HRV) from about 1600 to 1850, with an indication of our confidence in making each comparison. Stand characteristics are evaluated for areas modified by human activities only. Landscape characteristics are evaluated across the landscape for parts or all of the MBNF. For further discussion of each variable, refer to section 8 and the pages listed in the column on the right.

Variable	**Within HRV**	**Outside HRV**	**Confidence**	**Pages with discussion of variable**
Disturbances				
Stand-replacing fires	x		Moderate	24-29, 53-59, 62, 65-70, 72, 81-83, 85-86, 88-89, 93-94
Surface fires at low elevations on dry sites		x	High	65-70, 83-89
Insect outbreaks	x		High	30-33, 42, 70-72, 82, 85-87
Disease outbreaks, except for dwarf mistletoe in some areas	x	x *if white pine blister rust present*	Moderate	34-36, 72-73, 87, 103-104
Wind	x		High	36-38, 73
Stand structure at high elevations				
Density of trees in all age and size classes	x		High	23-39, 43-44
Tree cover and the density and size of canopy gaps		x	Moderate	23-39, 44-45
Understory plant density and cover	x		Moderate	45-46
Root/shoot ratios	x		High	46
Tree species and genetic diversity	x		High	46
Diversity of all plants	x		Moderate	47
Age and size-class structure of trees		x	Moderate	23-39, 47-49
Forest floor depth	x		Moderate	49-50
Mineral soil affected		x	High	50-51
Snag density		x	High	51
Coarse woody debris		x	High	51-53
Landscape features at high elevations				
Number and proportion of different land cover types	x		High	20-23, 53-58, 61
Forest/non-forest land area ratio	x		High	60
Rate at which new, uniformly distributed, mid-sized patches are formed		x	Moderate	23-39, 61, 99
Proportion of forest land in different stand-age classes		x	Moderate	58-59, 99

Proportion of land with high snag and CWD densities (young and old successional stages)		x	High	51-53, 61
Proportion of land in edge, interior forest, and patchy forest		x	High	53-58, 60
Proportion of land with stands of high tree density		x	Moderate	61
Stand structure at low elevations				
Species diversity	x		Low	80
Spatial distribution of trees	x		Low	78-79
Size- and age-class structure		x	Moderate	73-78
Understory plant composition and cover		x	Low	73-77, 80
Forest floor depth		x	High	80
Snag density		x	High	79-80
Coarse woody debris		x	High	79-80
Landscape features at low elevations				
Proportion of landscape in high tree density stands		x	Moderate	81-82
Proportion of landscape with old trees and high snag and CWD densities		x	Moderate	77-82
Interspersion and proportion of landscape in different cover types		x	Low	77-82
Aspen				
Representation in landscape	x		Moderate	83-89
Insect and disease abundance	x		Moderate	86-87
Proportion of stands with conifers in understory		x	Low	87-88
Exotic plant species cover		x	High	45, 75, 95-96, 101
Non-forest vegetation	*Inadequate information to assess HRV, but recovering from earlier periods of heavy grazing; exotic plants are common in some areas*			89-96, 101-102

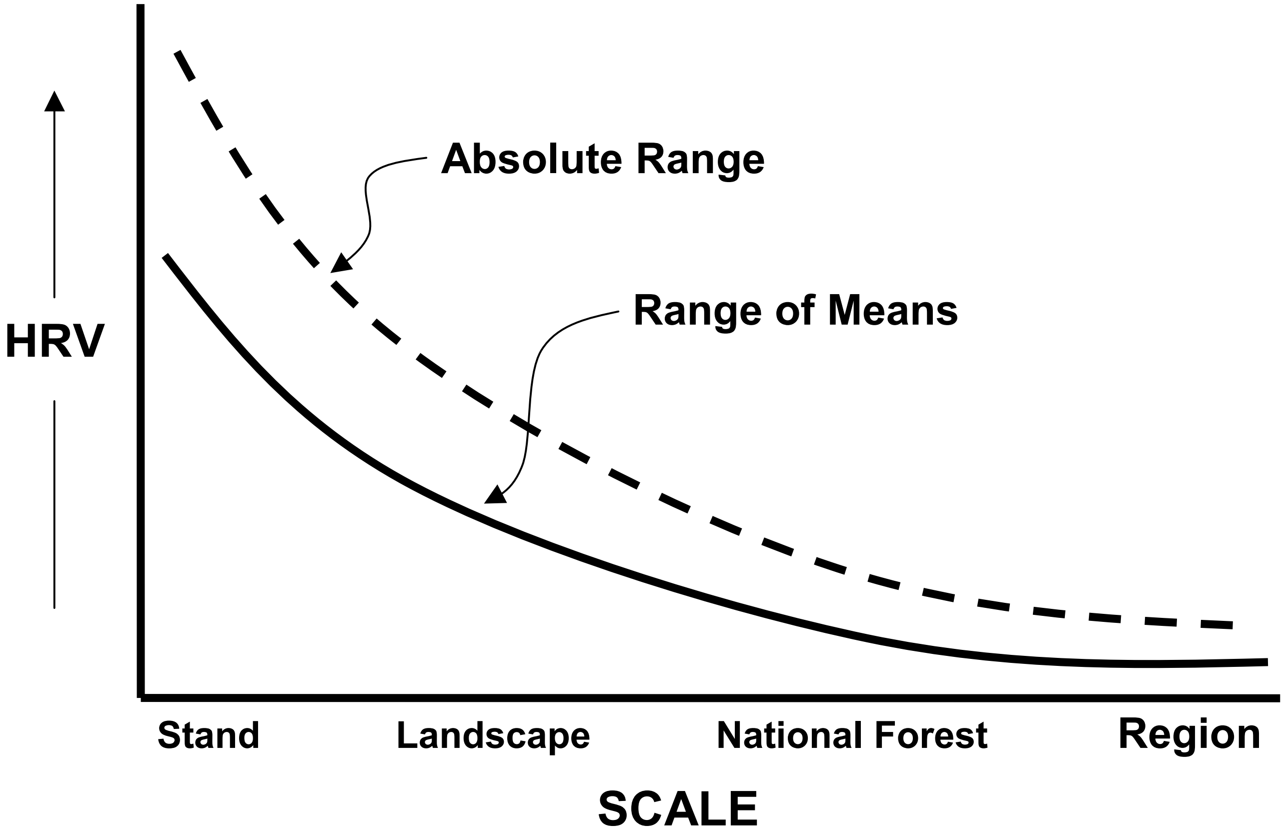


Figure 1. Illustration of the decline in the HRV as the spatial scale increases. Also, this figure illustrates how the HRV, estimated using the range of means (defined in Figure 2 and the text), is always less than the absolute range between the maximum and minimum for the entire period.

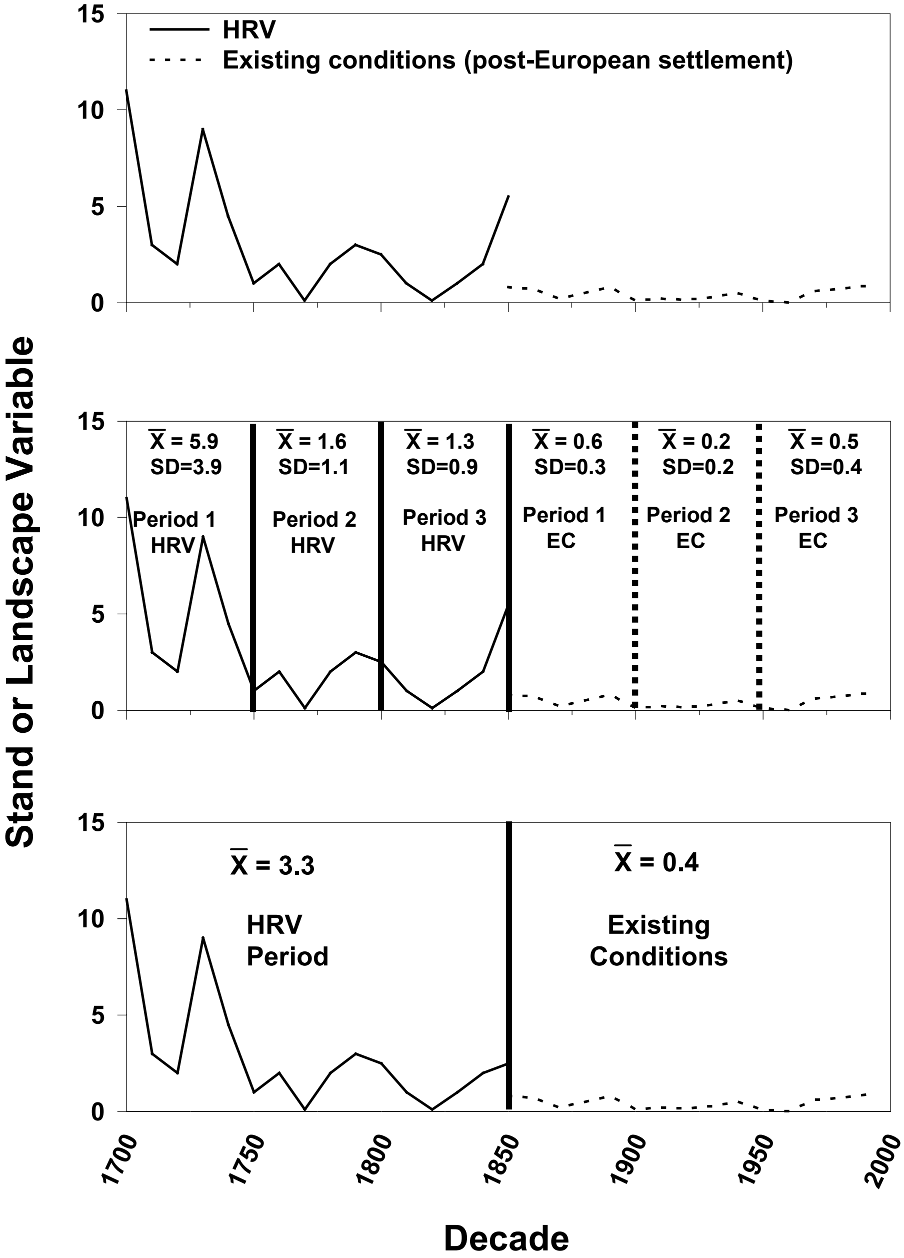


Figure 2. Three approaches for estimating the historic range of variability (HRV). Top: If HRV is defined as the absolute range for a variable, then the range of existing conditions (EC) in the example (0-0.8) falls within the HRV (0-11). Center: If HRV and EC are defined as the range of means ($\bar{x}$) for a variable over consecutive time periods, an approach which ignores extremes, the EC (0.2-0.6) falls outside the HRV (1.3-5.9). If HRV is the range of standard deviations of the variable, the EC (0.2-0.4) falls outside the standard deviation for the historic conditions (0.9-3.9). We used the range of means as the conceptual definition of HRV in this report. Bottom: Only one mean usually is reported for the historic period, and thus, no range of means can be determined. In such cases, the mean for existing conditions must be substantially higher or lower than the HRV mean to be considered a deviation from the HRV. In this example, the HRV would have to be less than ± 80% of the historic mean of 3.3 to have the EC mean (0.4) still fall outside the HRV (0.7-5.9) using the same data given in the center graph.

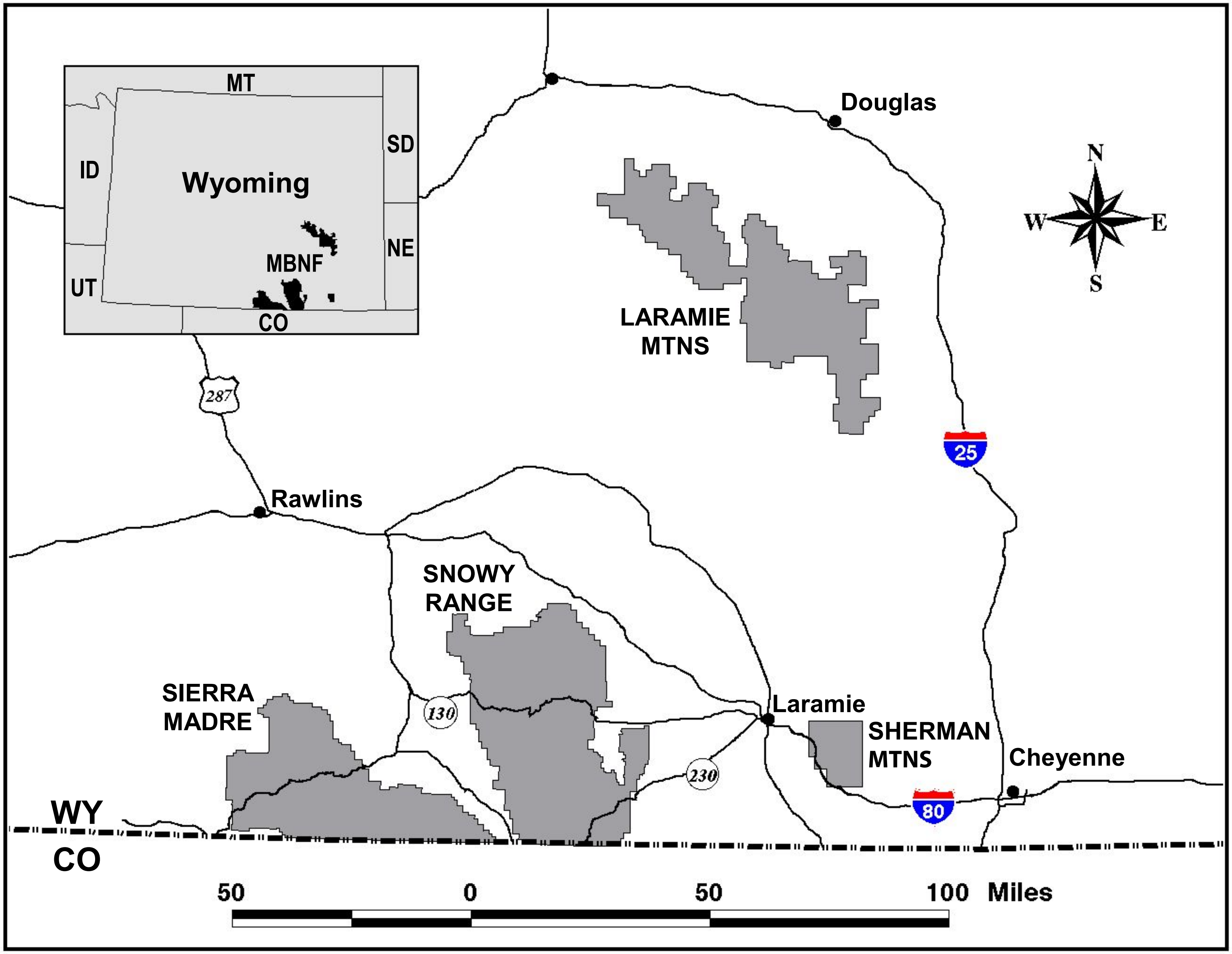
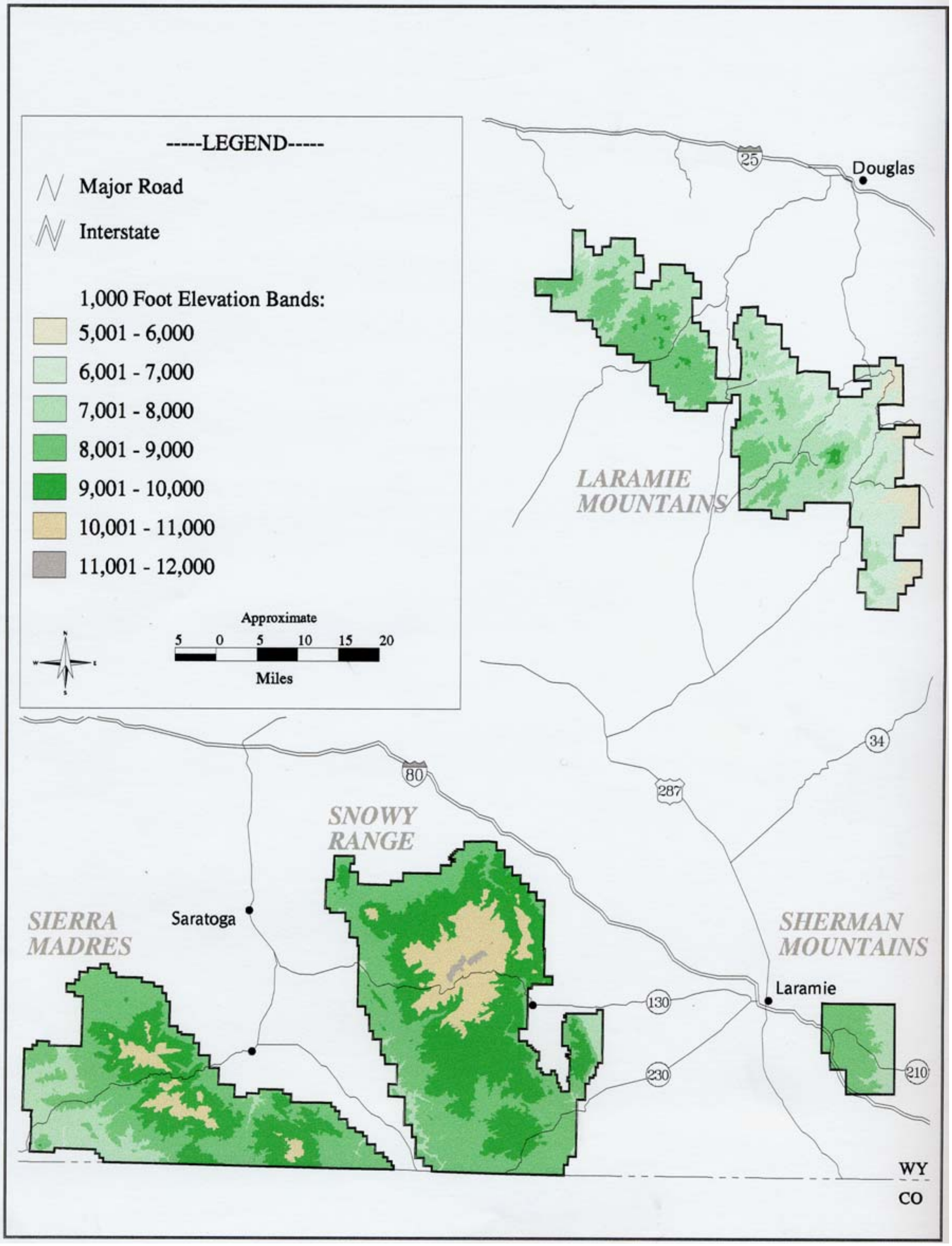

MT
SD
ID
Wyoming
NE
MBNF
UT
CO
Douglas
N
W
E
S
LARAMIE
MTNS
287
25
Rawlins
SNOWY
RANGE
SIERRA
MADRE
130
Laramie
SHERMAN
MTNS
230
Cheyenne
80
WY
CO
50
0
50
100 Miles

Figure 3. Location of the four units of the Medicine Bow National Forest in southeastern Wyoming. The Sherman Mountains unit is also known as the Pole Mountain unit, and the Laramie Mountains unit is sometimes referred to as the Laramie Peak unit.



-----LEGEND-----
Major Road
Interstate
1,000 Foot Elevation Bands:
5,001 - 6,000
6,001 - 7,000
7,001 - 8,000
8,001 - 9,000
9,001 - 10,000
10,001 - 11,000
11,001 - 12,000
Approximate
5 0 5 10 15 20
Miles
N
W
E
S
25
Douglas
LARAMIE MOUNTAINS
34
80
287
SNOWY RANGE
SIERRA MADRES
Saratoga
SHERMAN MOUNTAINS
Laramie
130
230
210
WY
CO

Figure 4. Map of elevation zones for the four units that comprise the Medicine Bow National Forest. Source: MBNF

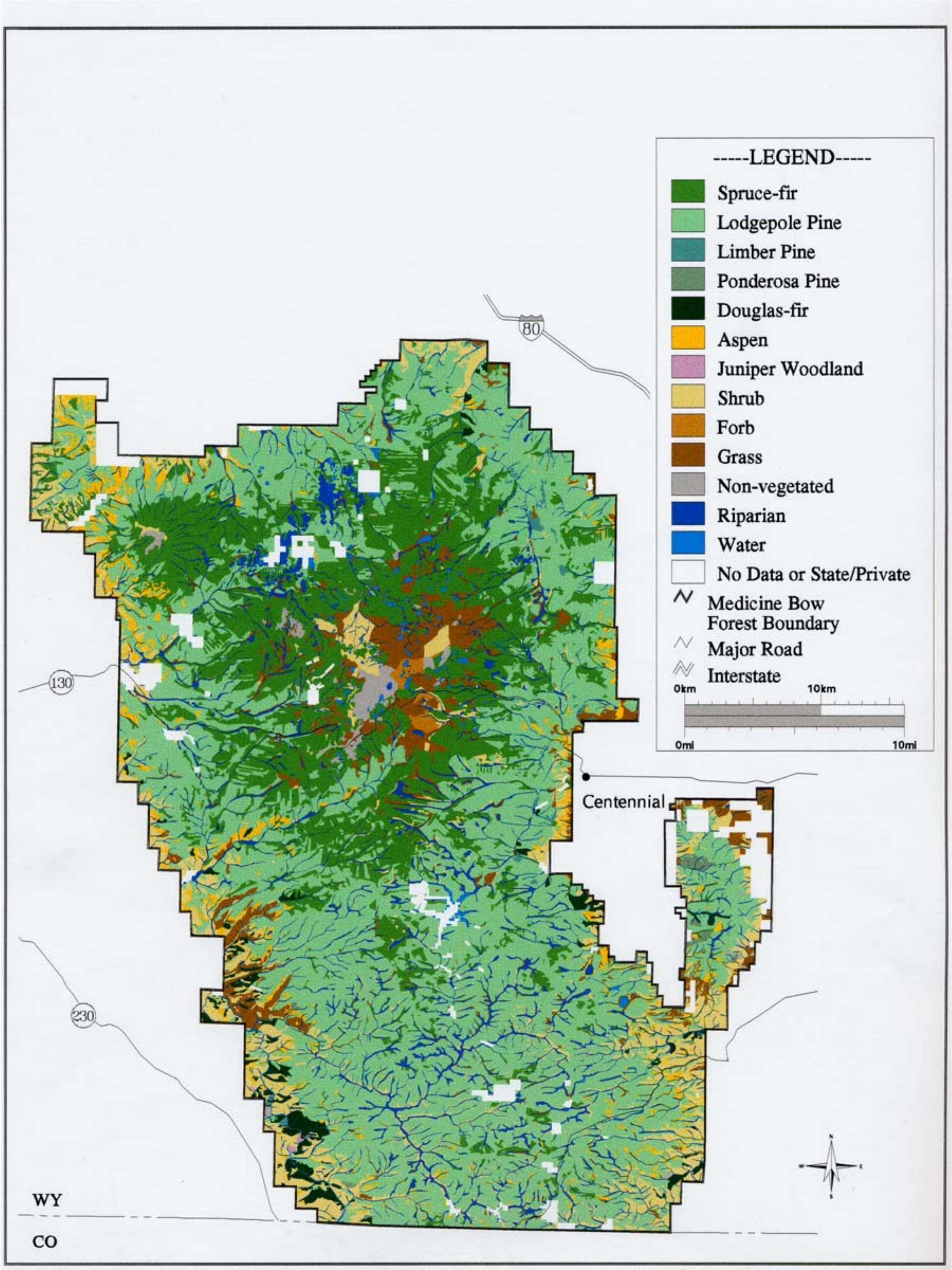

-----LEGEND-----
Spruce-fir
Lodgepole Pine
Limber Pine
Ponderosa Pine
Douglas-fir
Aspen
Juniper Woodland
Shrub
Forb
Grass
Non-vegetated
Riparian
Water
No Data or State/Private
Medicine Bow Forest Boundary
Major Road
Interstate
0km
10km
0mi
10mi
80
130
230
Centennial
WY
CO
N
E
S
W

Figure 5a. Land cover map for the Snowy Range unit of the MBNF. Source: MBNF.

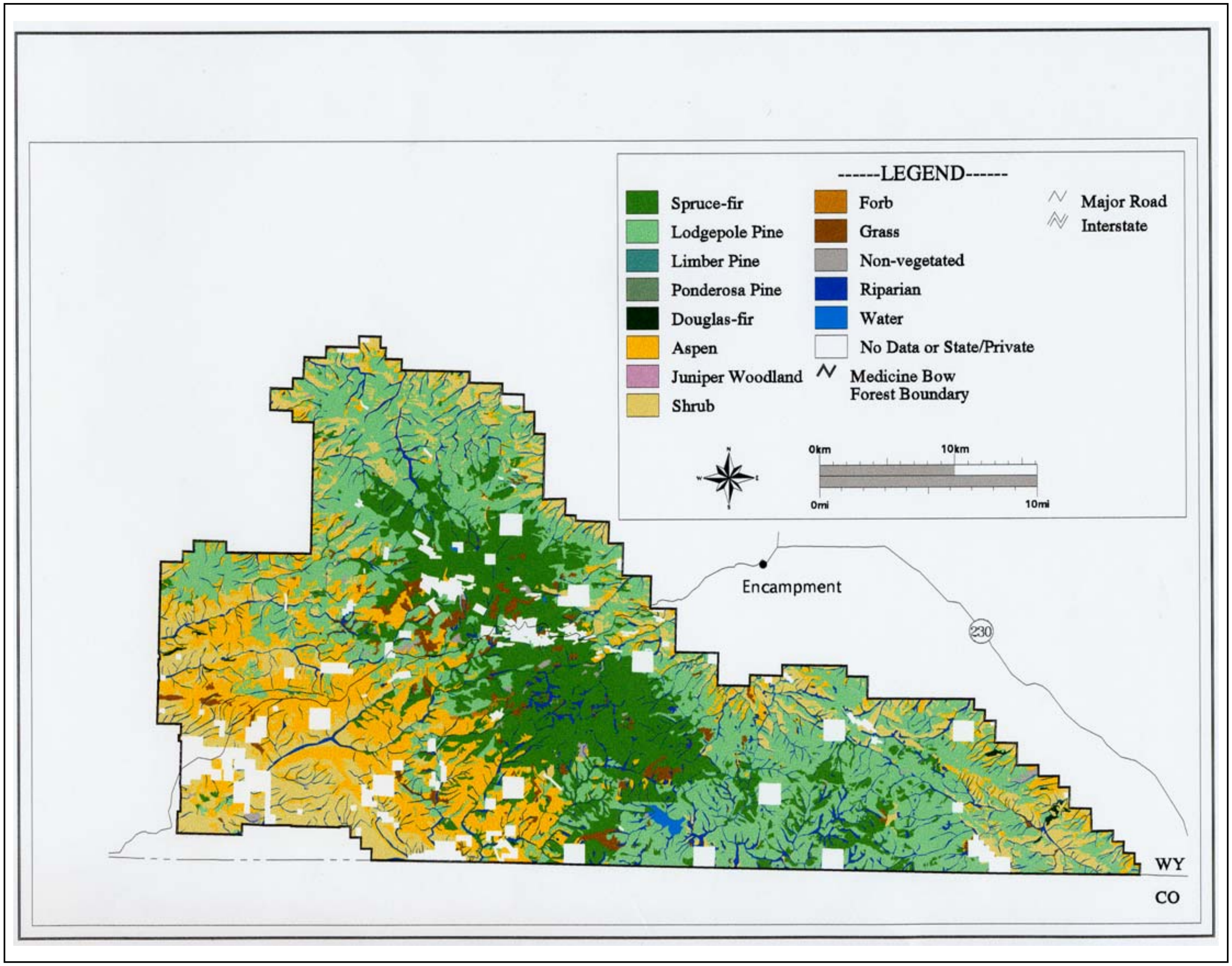
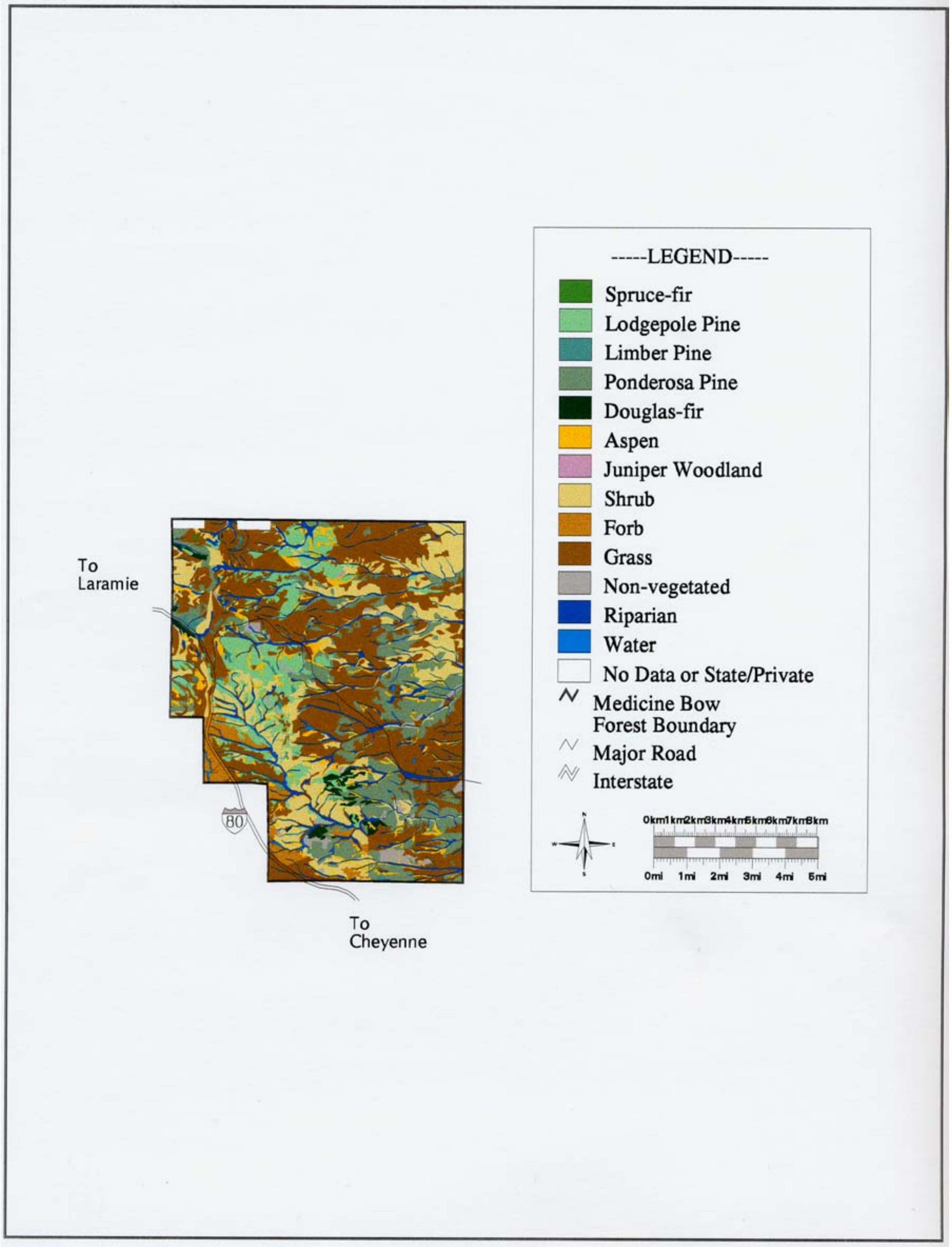

------LEGEND------
Spruce-fir
Lodgepole Pine
Limber Pine
Ponderosa Pine
Douglas-fir
Aspen
Juniper Woodland
Shrub
Forb
Grass
Non-vegetated
Riparian
Water
No Data or State/Private
Medicine Bow Forest Boundary
Major Road
Interstate
0km
10km
0mi
10mi
Encampment
230
WY
CO

Figure 5b. Land cover map for the Sierra Madre unit of the MBNF. Source: MBNF.



-----LEGEND-----
Spruce-fir
Lodgepole Pine
Limber Pine
Ponderosa Pine
Douglas-fir
Aspen
Juniper Woodland
Shrub
Forb
Grass
Non-vegetated
Riparian
Water
No Data or State/Private
Medicine Bow Forest Boundary
Major Road
Interstate
0km 1km 2km 3km 4km 5km 6km 7km 8km
0mi 1mi 2mi 3mi 4mi 5mi
N
W
E
S
To Laramie
80
To Cheyenne

Figure 5c. Land cover map for the Sherman Mountains unit of the MBNF. Source: MBNF.

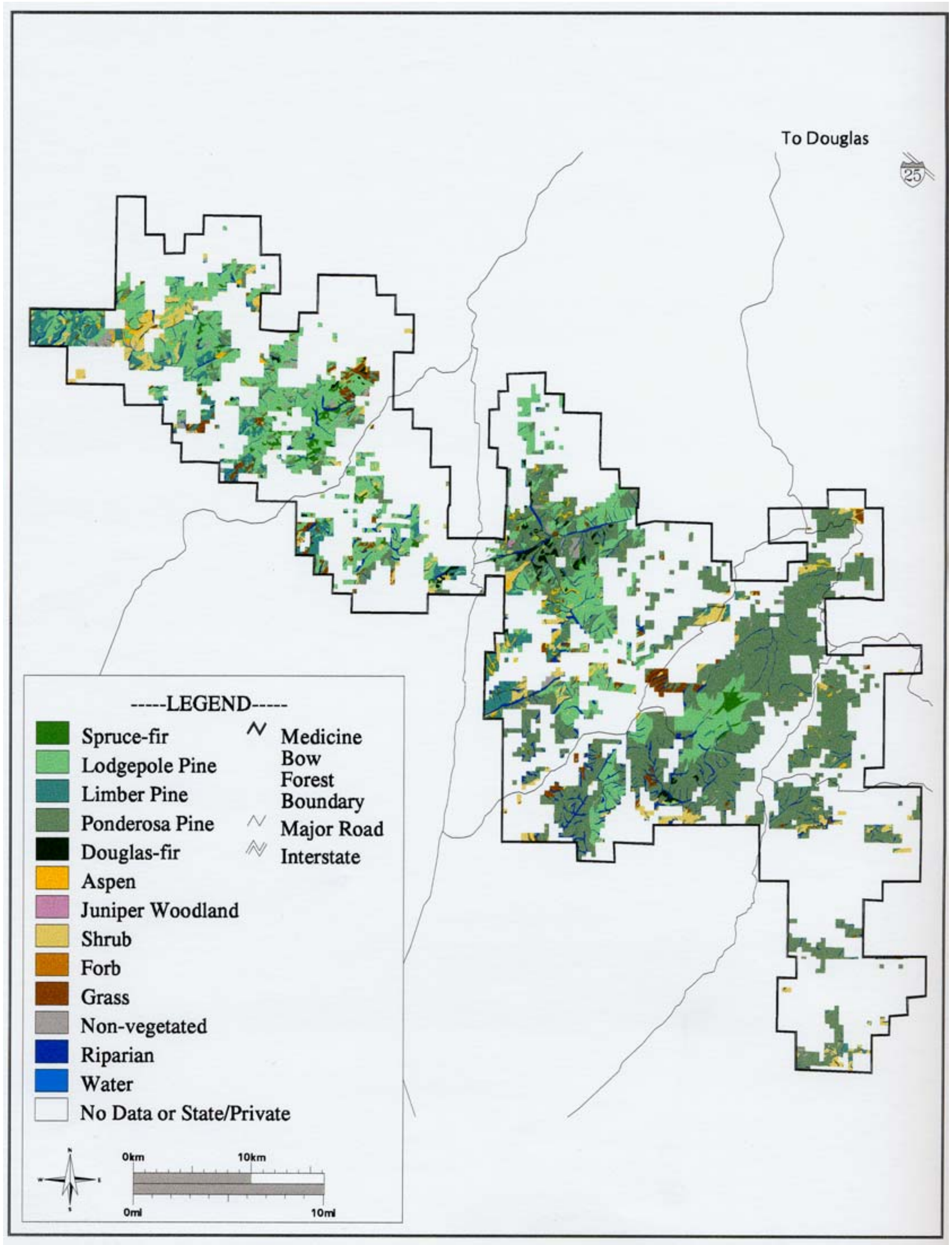
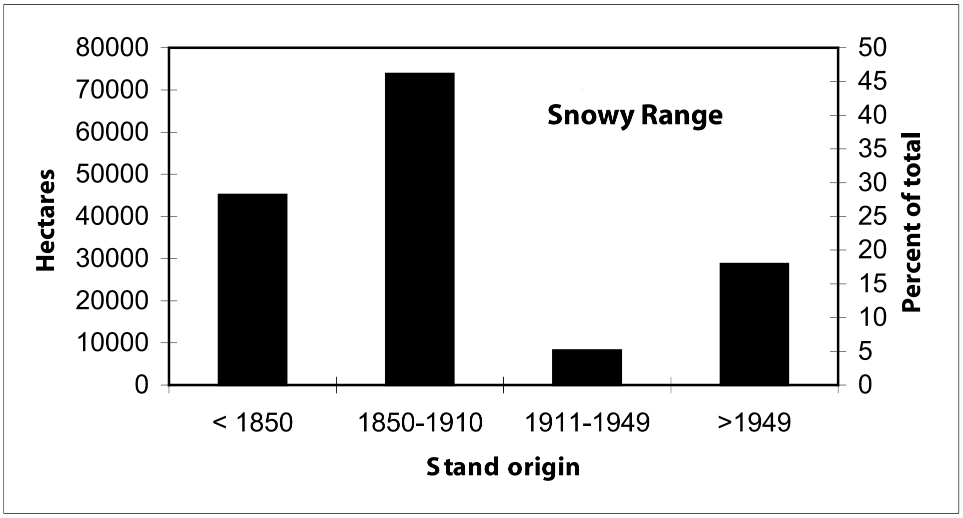

To Douglas
25
-----LEGEND-----
Spruce-fir
Lodgepole Pine
Limber Pine
Ponderosa Pine
Douglas-fir
Aspen
Juniper Woodland
Shrub
Forb
Grass
Non-vegetated
Riparian
Water
No Data or State/Private
Medicine Bow Forest Boundary
Major Road
Interstate
0km
10km
0mi
10mi

Figure 5d. Land cover map for the Laramie Mountains unit of the MBNF. Source: MBNF.



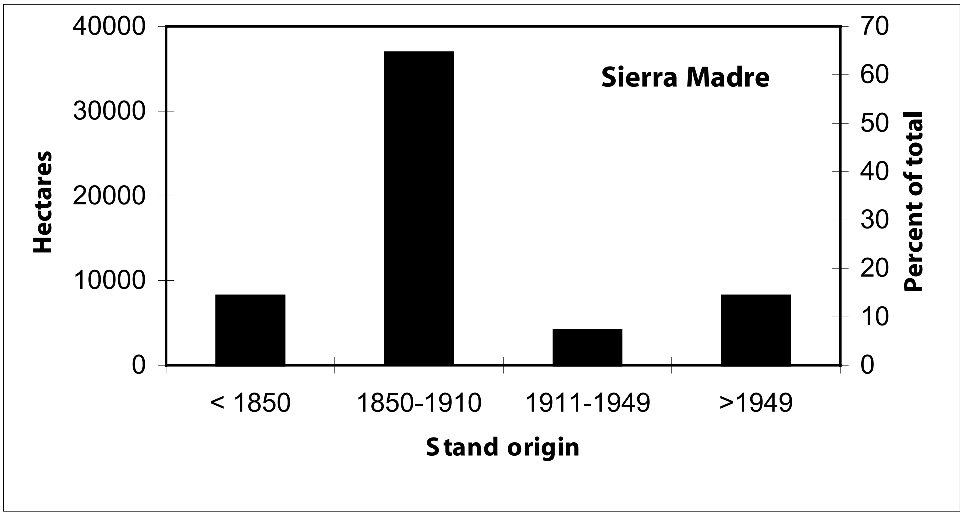


Figure 6. Age distribution of lodgepole pine and spruce fir forests on the Snowy Range and Sierra Madre units of the MBNF. Only areas for which age data are available are included (37% of the lodgepole pine and spruce-fir forests on the Snowy Range; 32% on the Sierra Madre). Source: MBNF-RIS; adapted from von Ahlefeldt and Speas (1996).

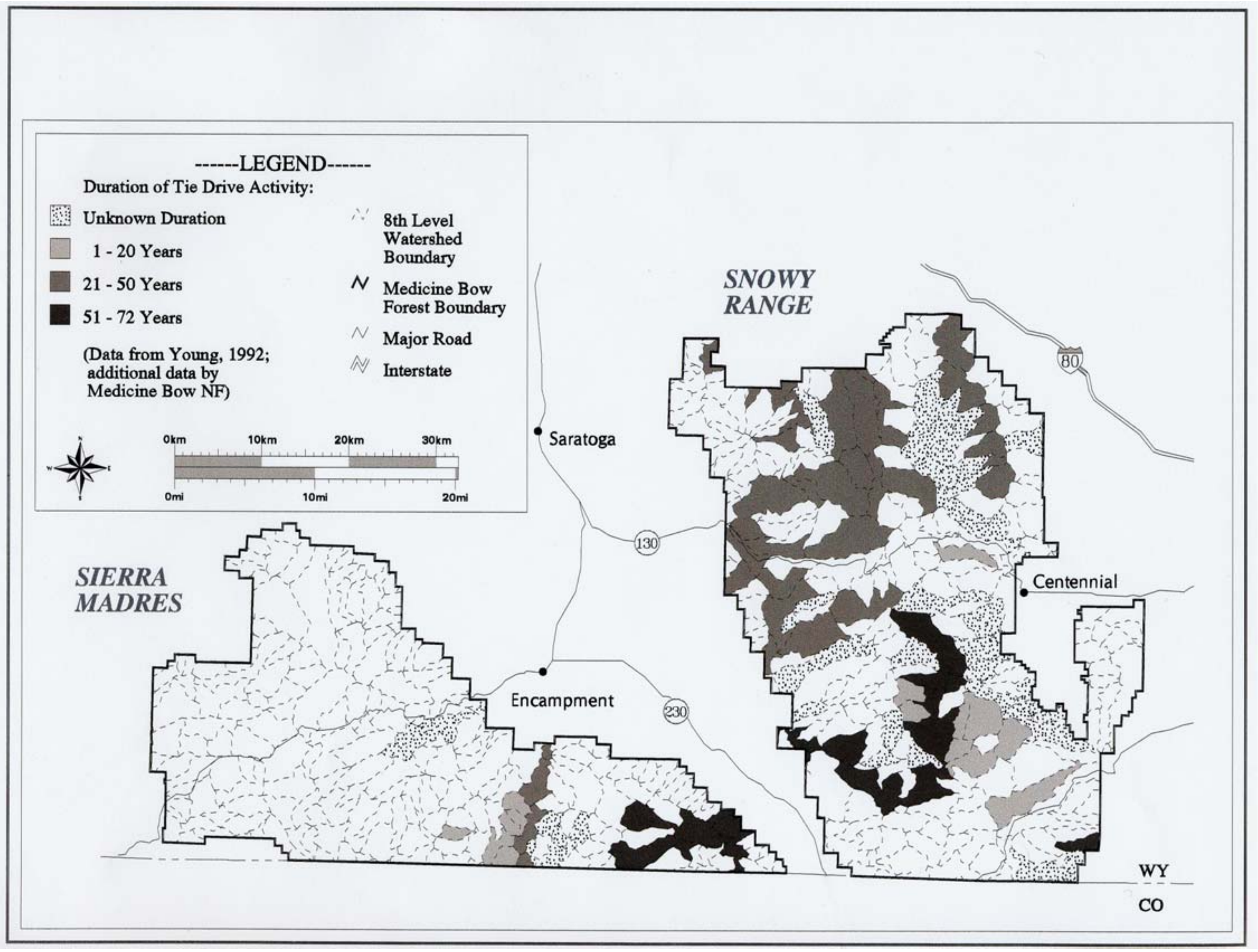

------LEGEND------
Duration of Tie Drive Activity:
Unknown Duration
1 - 20 Years
21 - 50 Years
51 - 72 Years
(Data from Young, 1992; additional data by Medicine Bow NF)
8th Level Watershed Boundary
Medicine Bow Forest Boundary
Major Road
Interstate
0km
10km
20km
30km
0mi
10mi
20mi
SNOWY RANGE
SIERRA MADRES
Saratoga
Encampment
Centennial
130
230
80
WY
CO

Figure 7. Watersheds with streams known to have been used for trailroad tie drives on the Medicince Bow National Forest. Source: MBNF

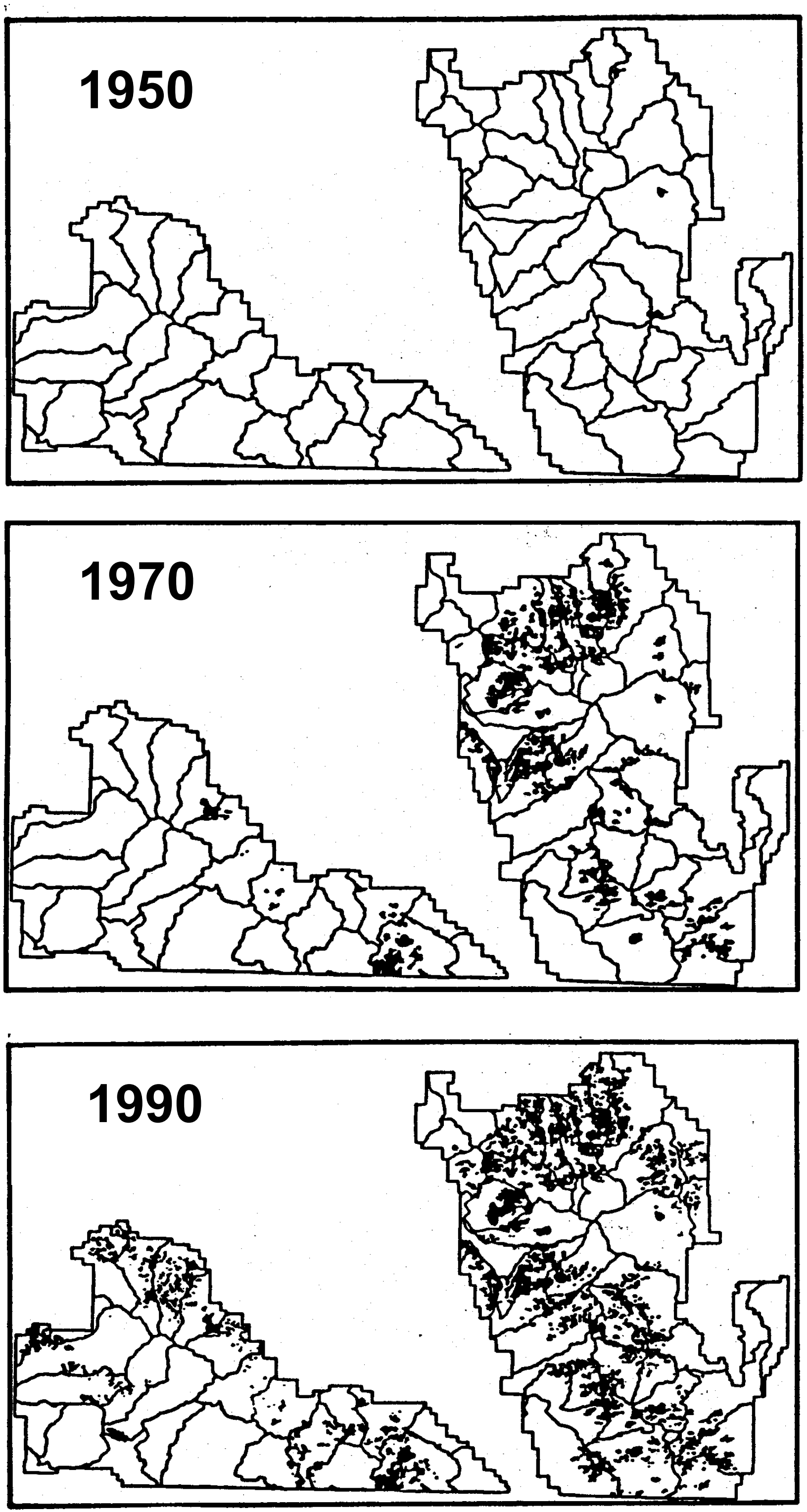

1950
1970
1990

Figure 8a. Cumulative land area affected by clearcut harvesting in 1950, 1970, and 1990. From Baker 1994.

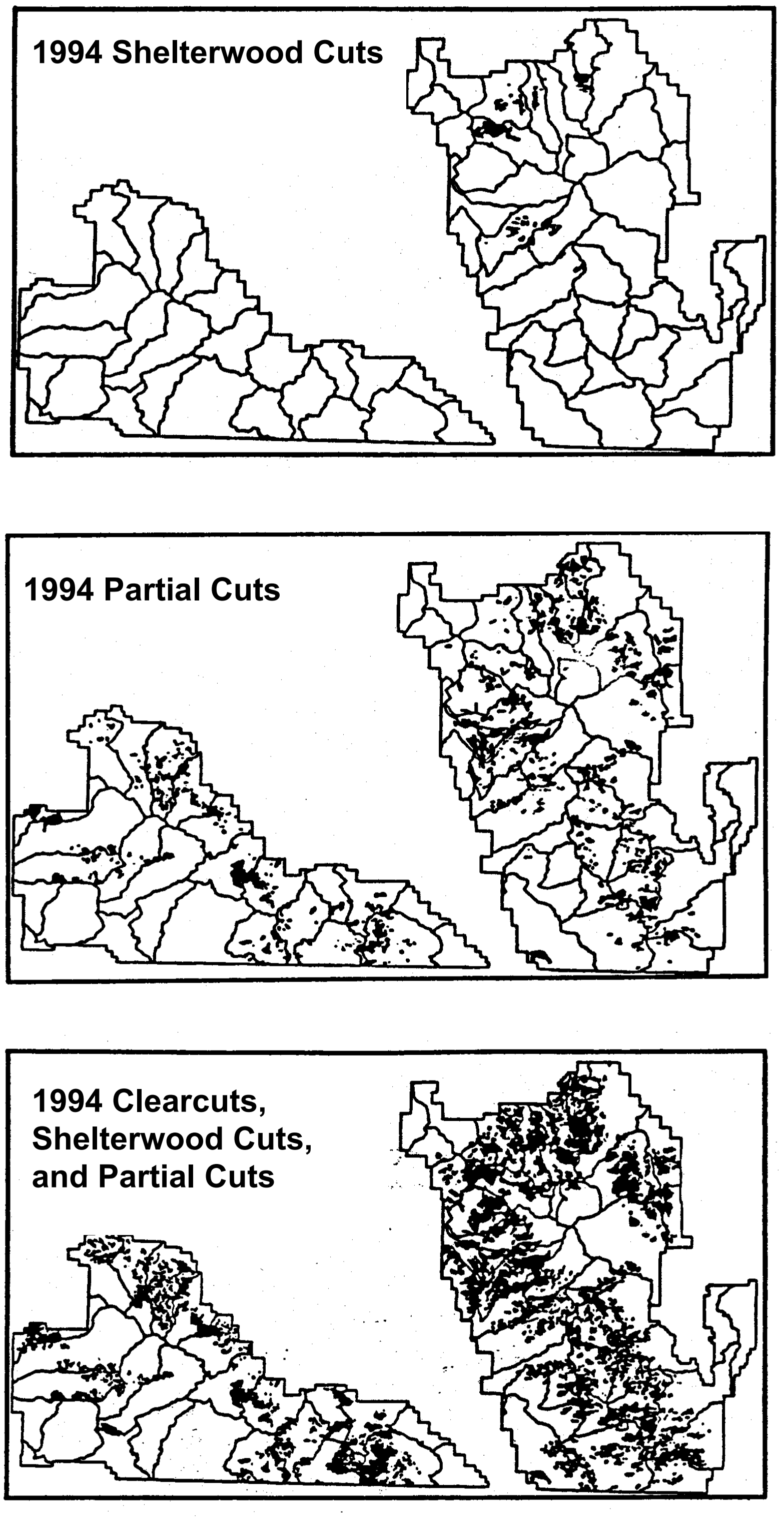

1994 Shelterwood Cuts
1994 Partial Cuts
1994 Clearcuts, Shelterwood Cuts, and Partial Cuts

Figure 8b. Cumulative land area affected directly by shelterwood harvesting by 1994, partial cuts by 1994, and all clearcuts, shelterwood cuts, and partial cuts by 1994. The depth of edge effect is not shown. From Baker 1994.

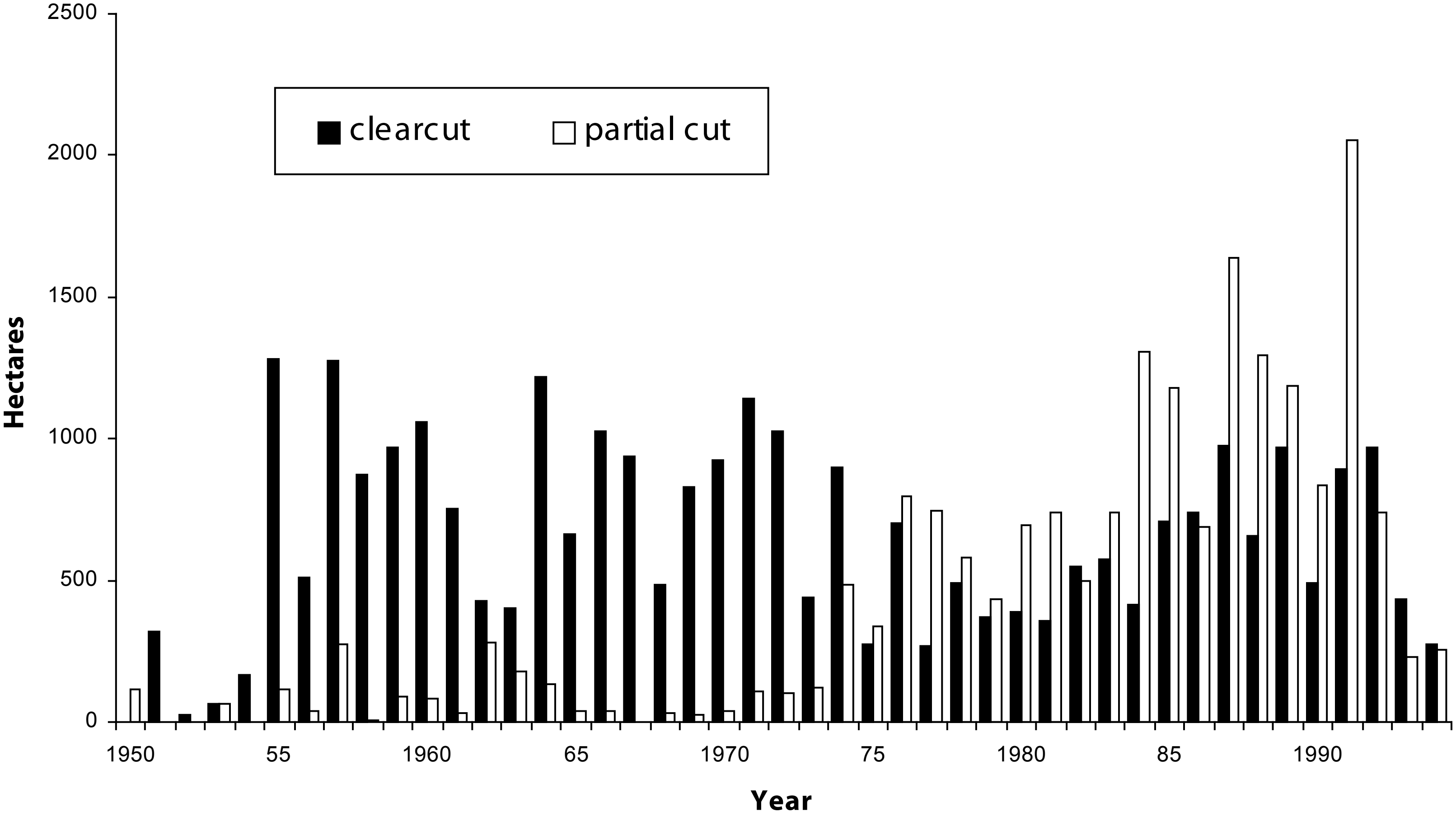


Figure 9. The land area in the Snowy Range and Sierra Madre where clearcutting and partial cutting have occurred during the period 1950-1994. Source: MBNF-RIS

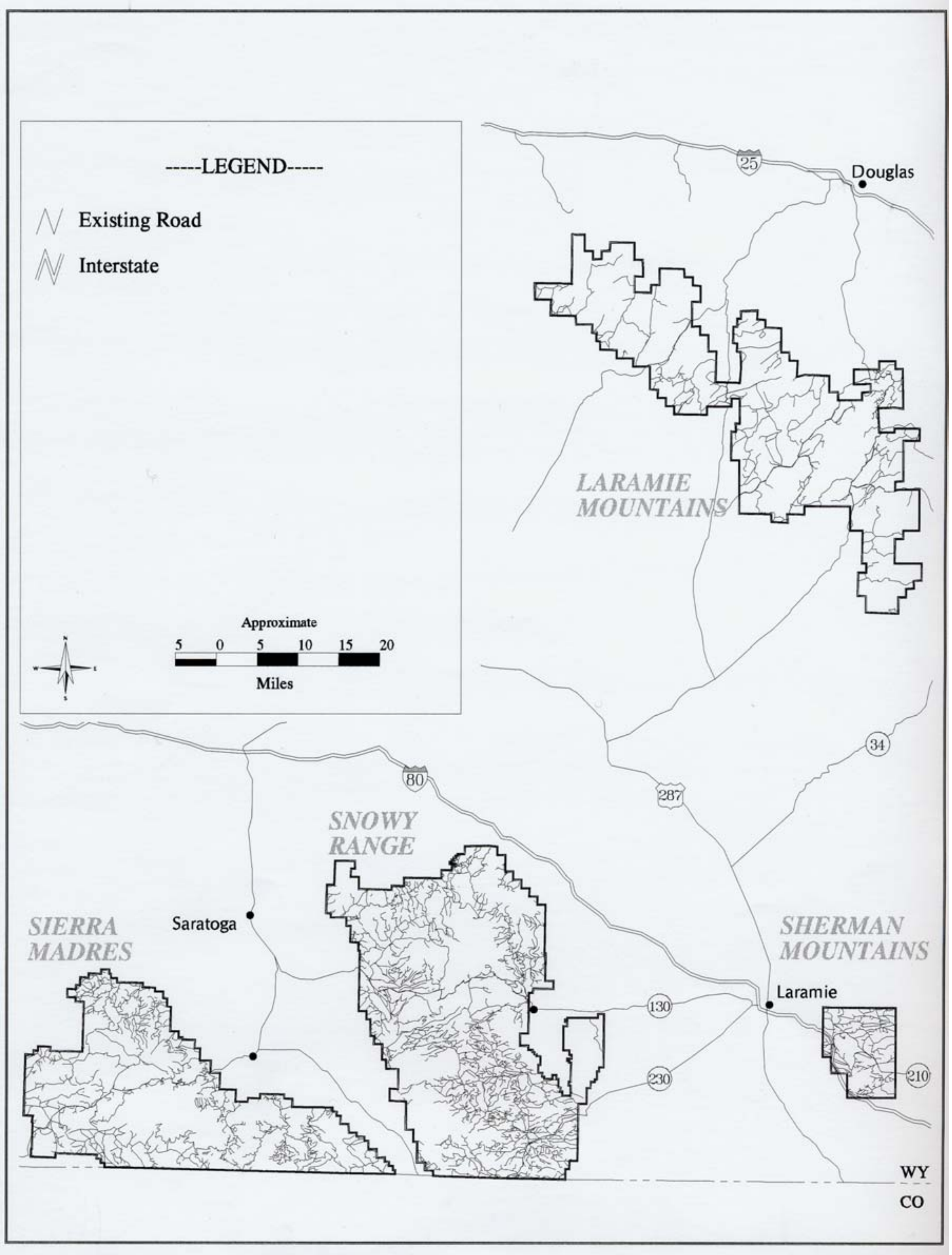

-----LEGEND-----
Existing Road
Interstate
Approximate
5 0 5 10 15 20
Miles
N
W
E
S
25
Douglas
LARAMIE MOUNTAINS
34
287
80
SNOWY RANGE
SIERRA MADRES
Saratoga
SHERMAN MOUNTAINS
Laramie
130
230
210
WY
CO

Figure 10. Roads within each of the four units of the Medicine Bow National Forest. All roads that exist as a feature of the present-day landscape are shown, including closed, abandoned, obliterated, and non-USFS roads. Road density in the MBNF is about 1.29 miles per square mile of national forest.

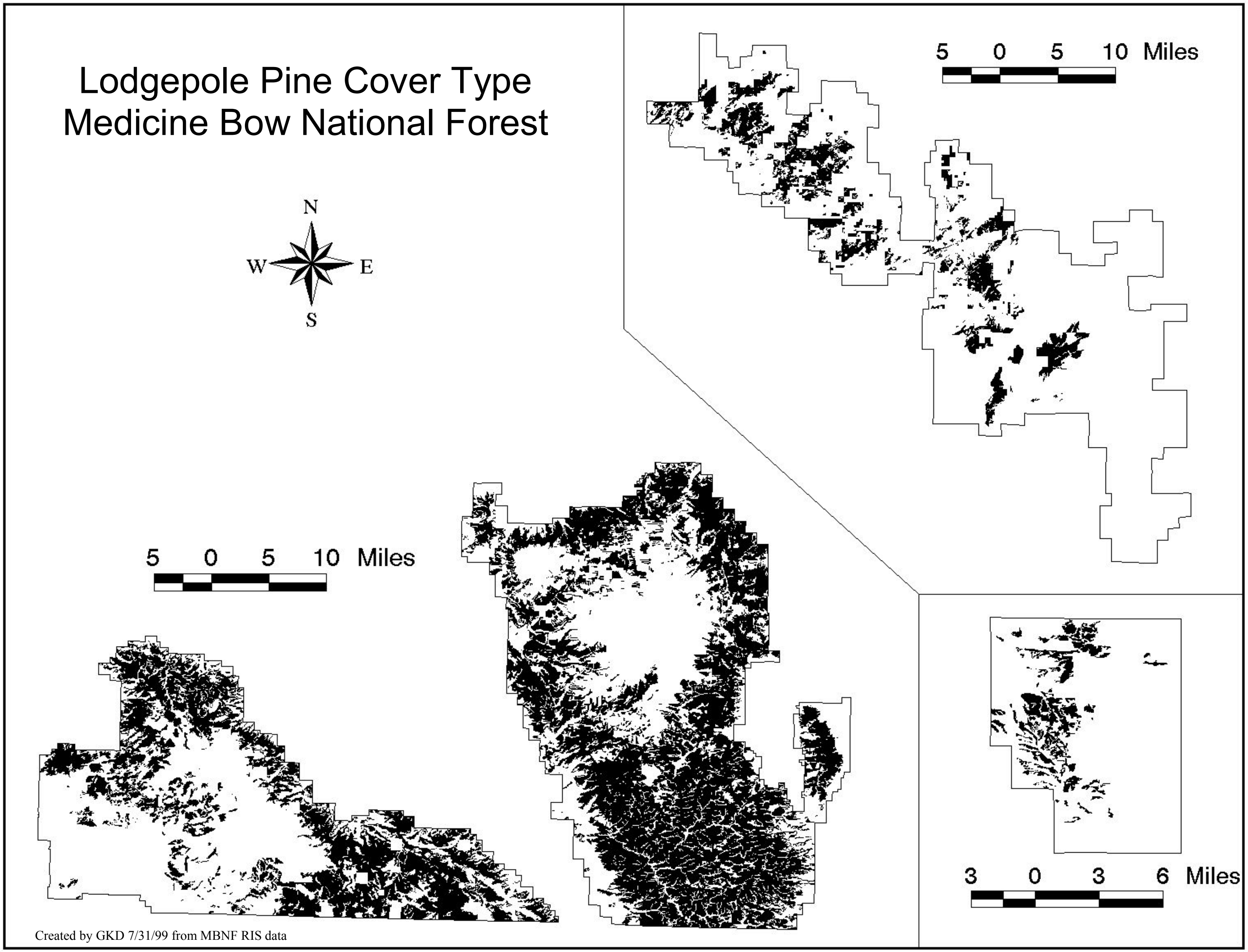

Lodgepole Pine Cover Type
Medicine Bow National Forest
N
W
E
S
5 0 5 10 Miles
5 0 5 10 Miles
3 0 3 6 Miles
Created by GKD 7/31/99 from MBNF RIS data

Figure 11. Distribution of lodgepole pine forests on the Medicine Bow National Forest. Source: MBNF

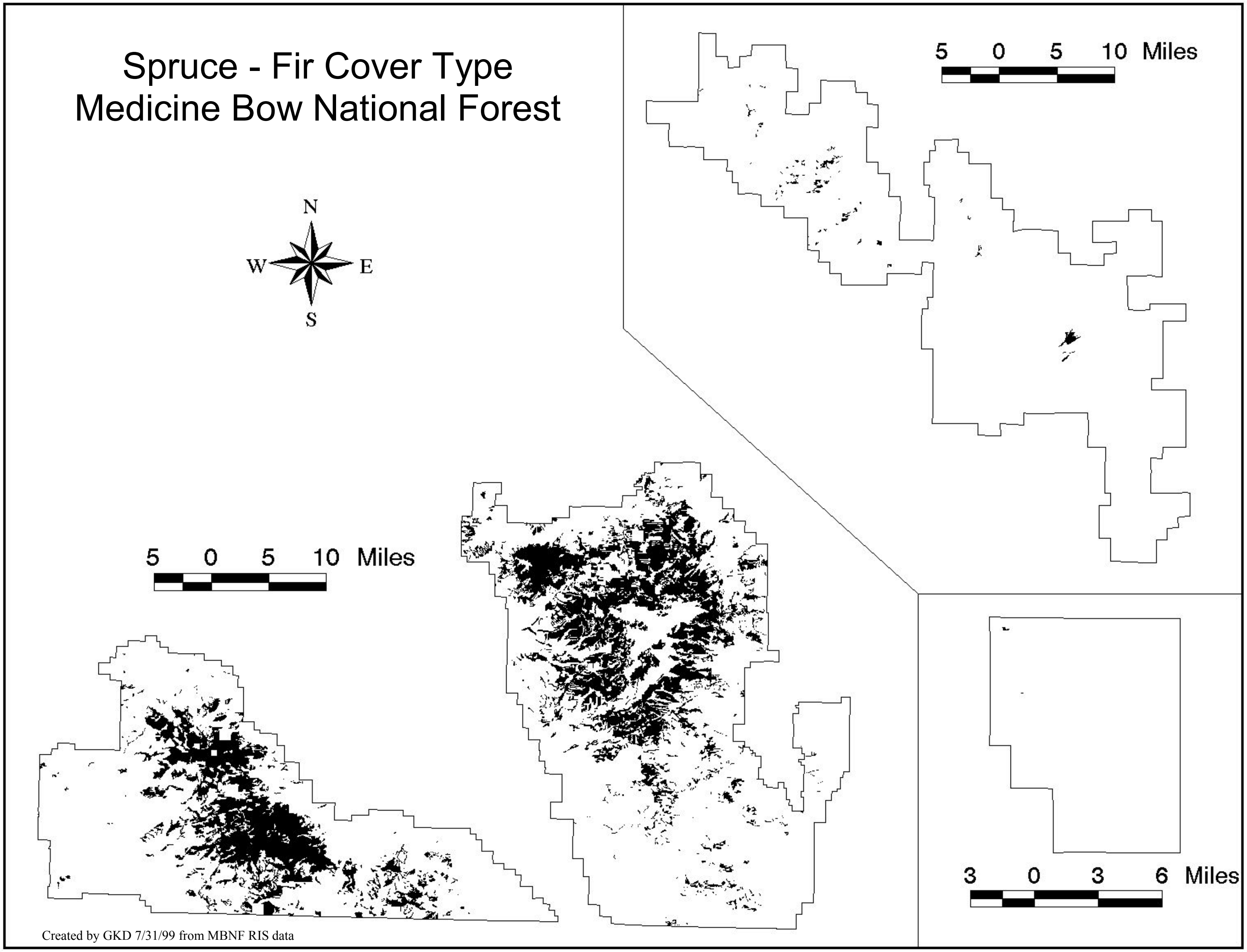

Spruce - Fir Cover Type
Medicine Bow National Forest
N
W
E
S
5 0 5 10 Miles
5 0 5 10 Miles
3 0 3 6 Miles
Created by GKD 7/31/99 from MBNF RIS data

Figure 12. Distribution of spruce–fir forests on the Medicine Bow National Forest. Source: MBNF

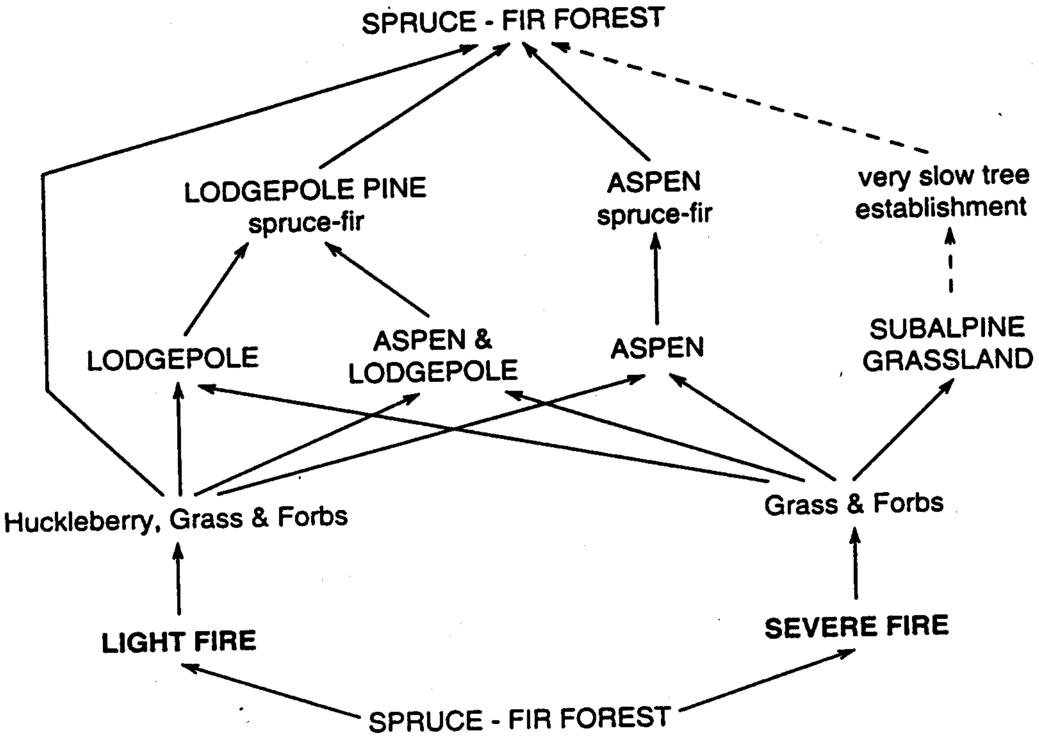

SPRUCE - FIR FOREST
LODGEPOLE PINE
spruce-fir
ASPEN
spruce-fir
very slow tree
establishment
LODGEPOLE
ASPEN &
LODGEPOLE
ASPEN
SUBALPINE
GRASSLAND
Huckleberry, Grass & Forbs
Grass & Forbs
LIGHT FIRE
SEVERE FIRE
SPRUCE - FIR FOREST

Figure 13. Alternative successional pathways following light and severe fires in spruce-fir forests of the Central Rocky Mountains. Note that spruce and fir are sometimes pioneer species, along with lodgepole pine or aspen, and that severe fires can lead to the establishment of grasslands or meadows at high elevations. Adapted from Stahelin 1943.

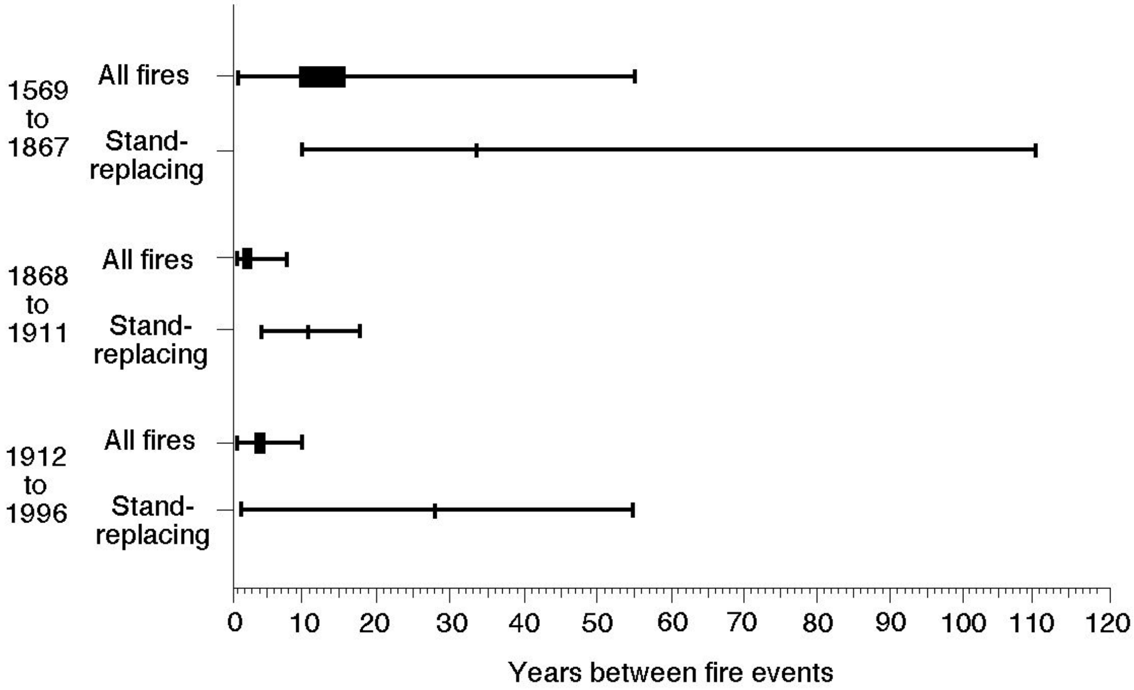


Figure 14. Mean fire intervals (MFI) for a portion of the Snowy Range as calculated by Kipfmueller (1997). Values represent the time between fires anywhere within the 3,241 ha (8,005 acre) study area that Kipfmueller used. Lines represent the range between minimum and maximum intervals observed, while the mark on each line represents the estimated MFI. Estimates for the pre-1867 fires are less precise.

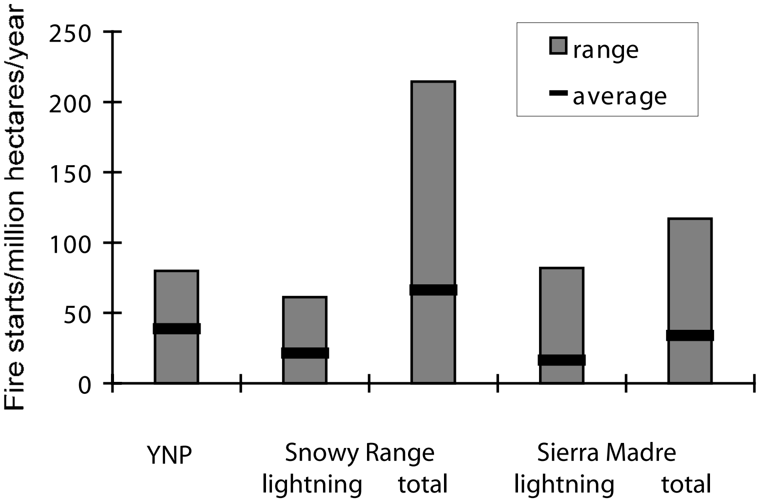


Fig. 15. Number of fire starts per year per million hectares in lodgepole pine and spruce-fir forests on the MBNF, compared to comparable data for Yellowstone National Park. MBNF data are for 1945-1993 (von Ahlefeldt and Speas 1996); YNP data are for 1972-1988 (Despain and Romme 1991).

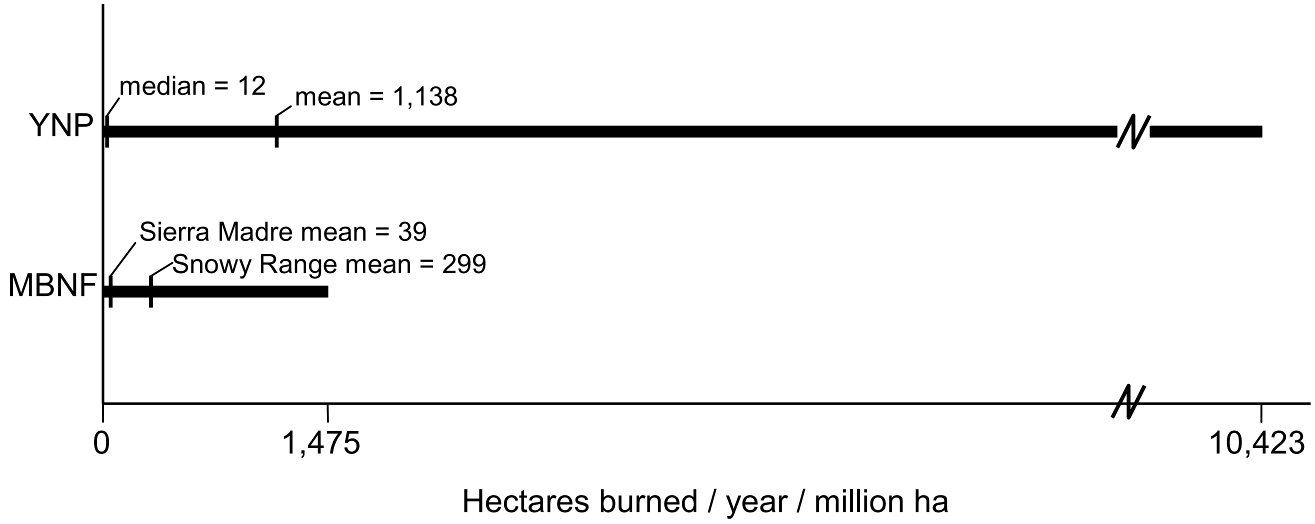


Figure 16. Hectares burned per year per million hectares in Yellowstone National Park and the Snowy Range and Sierra Madre portion of the Medicine Bow National Forest. The lines indicate the range. Also shown are the mean and median for YNP and the mean values on the MBNF. Data for YNP are from Despain and Romme (1991) and represent fire dynamics with no or little, in the area they studied, for the period 1972-87. Data for the MBNF were compiled by von Ahlefeldt and Speas (1996) and are for the period 1945-93, when attempts at fire suppression were common and often successful.

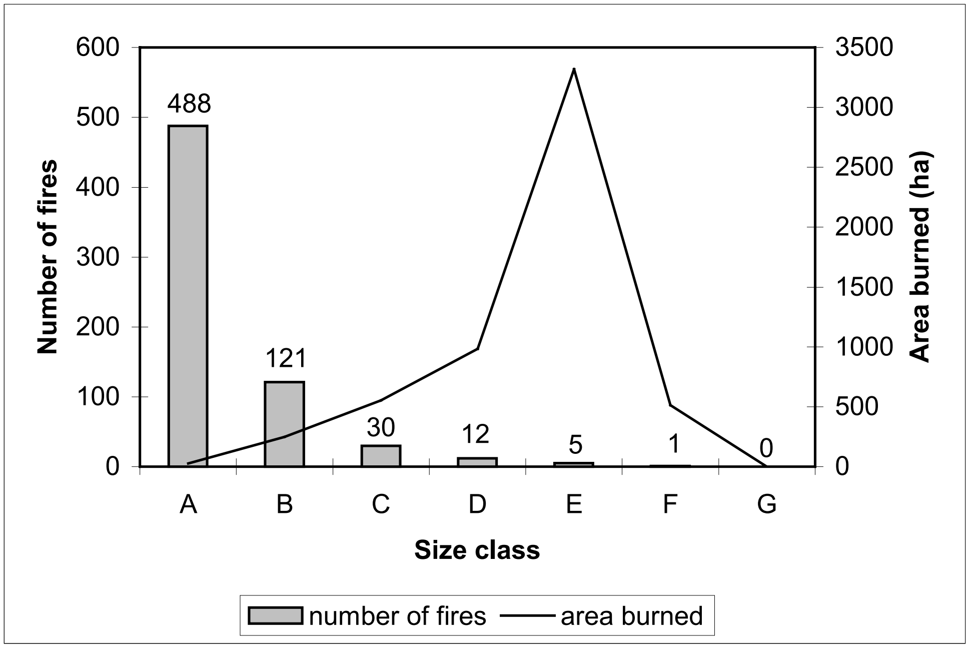


Figure 17. Number of fires and area burned on the Snowy Range and Sierra Madre, 1945-1993, based on data summarized by von Ahlefeldt and Speas (1996). Size classes are: A, < 0.1 ha; B, 0.1-3.9 ha; C, 4-39 ha; D, 40-122 ha; E, 123-409 ha; F, 410-2049 ha; G, >2049 ha.

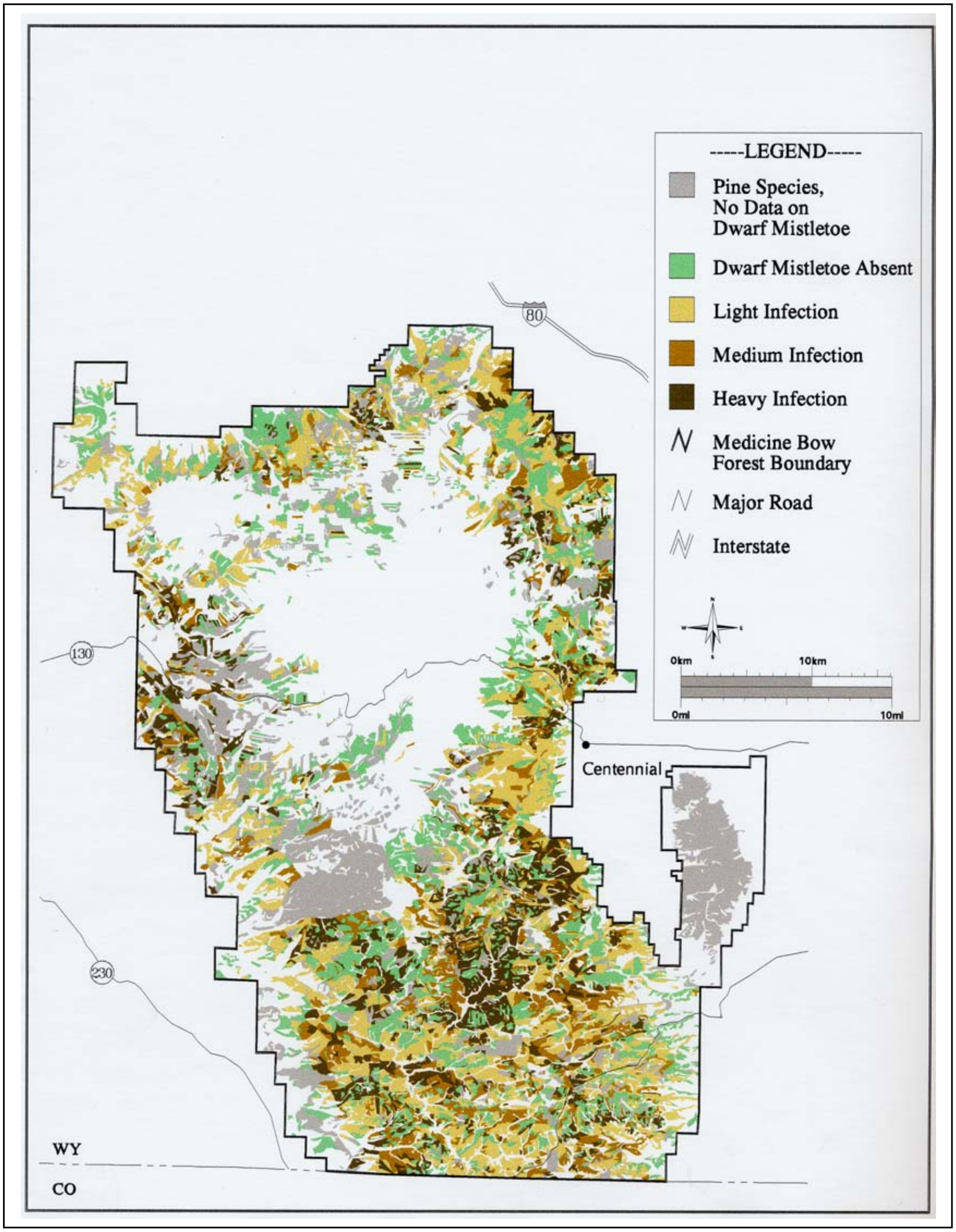
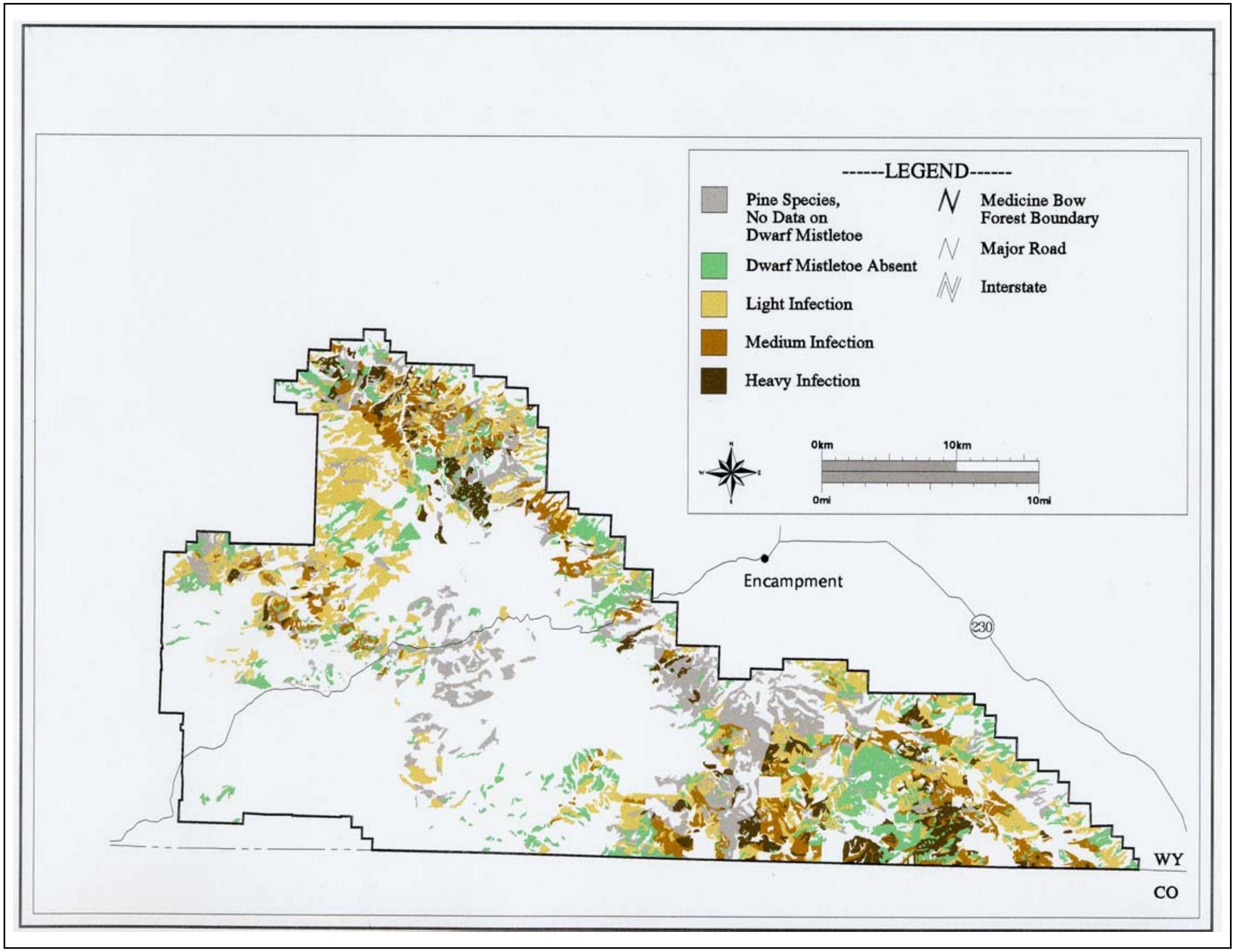

-----LEGEND-----
Pine Species, No Data on Dwarf Mistletoe
Dwarf Mistletoe Absent
Light Infection
Medium Infection
Heavy Infection
Medicine Bow Forest Boundary
Major Road
Interstate
0km
10km
0mi
10mi
80
130
230
Centennial
WY
CO

Figure 18a. Distribution of varying levels of mistletoe infection in the Snowy Range unit of the MBNF. Source: MBNF.



------LEGEND------
Pine Species, No Data on Dwarf Mistletoe
Dwarf Mistletoe Absent
Light Infection
Medium Infection
Heavy Infection
Medicine Bow Forest Boundary
Major Road
Interstate
0km
10km
0mi
10mi
Encampment
230
WY
CO

Figure 18b. Distribution of varying levels of mistletoe infection in the Sierra Madre unit of the MBNF. Source: MBNF

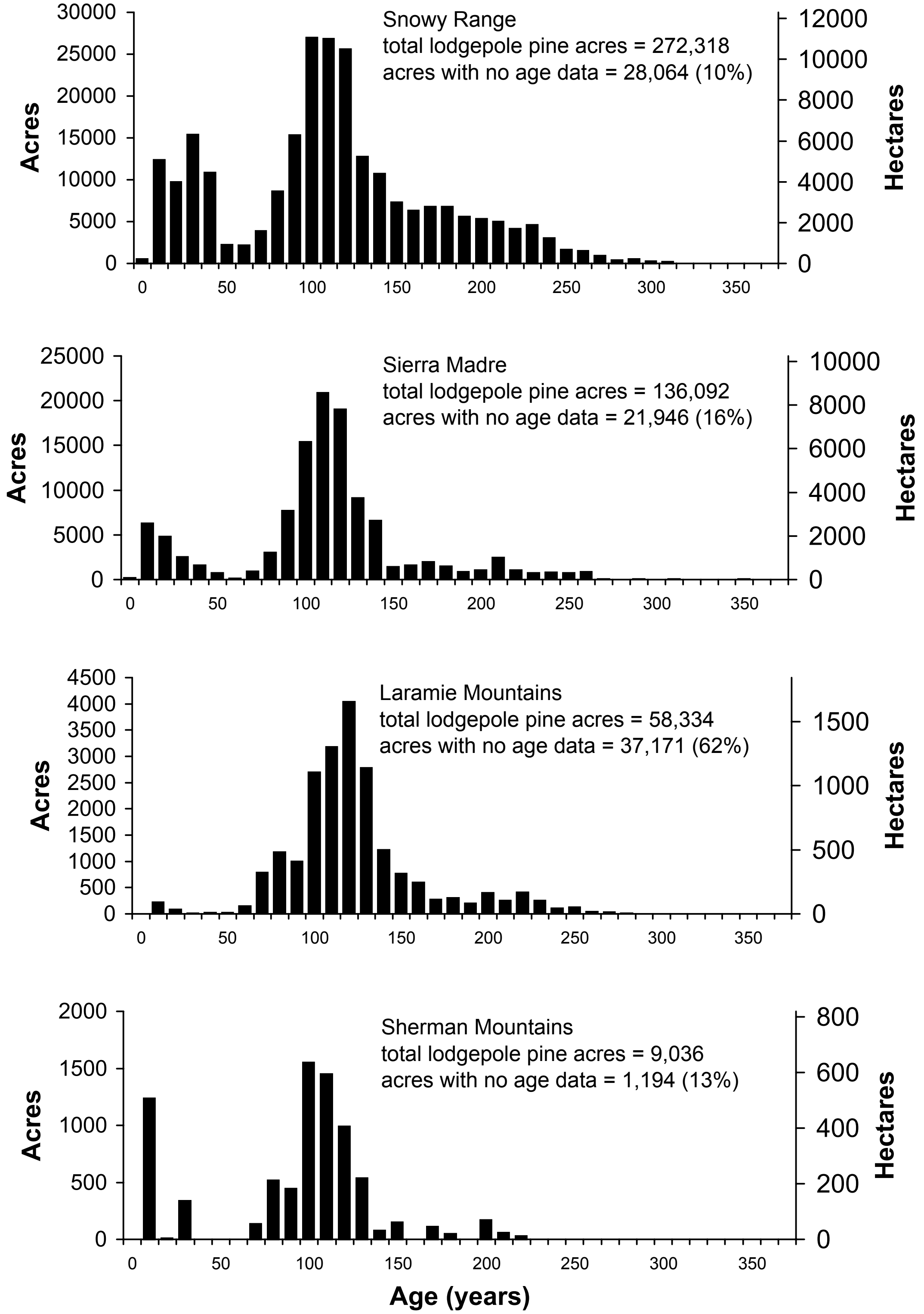
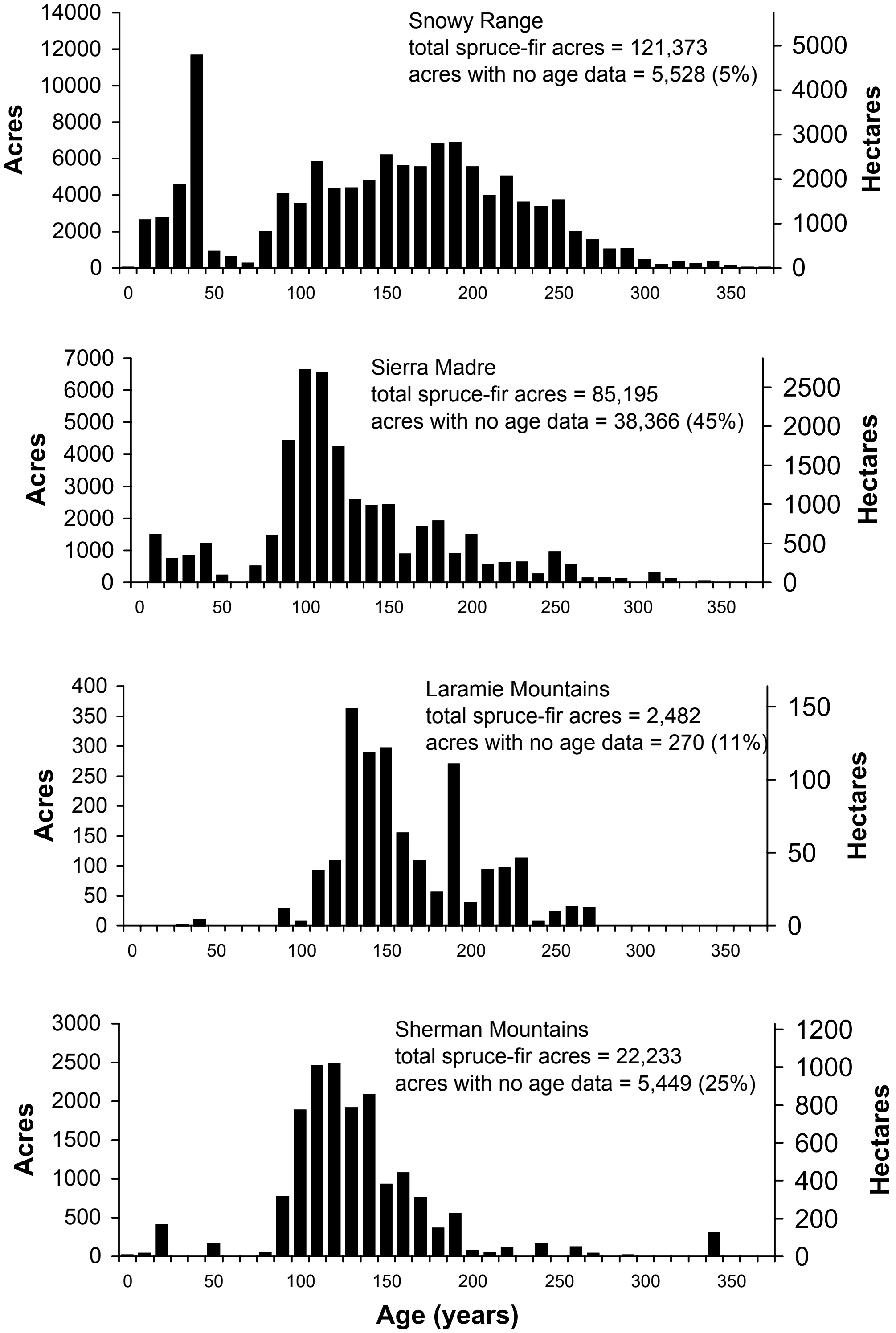


Figure 19a. Age distribution of lodgepole pine forests on the four units of the Medicine Bow National Forest. Bars represent ten-year age classes. Source: MBNF RIS


Snowy Range
total spruce-fir acres = 121,373
acres with no age data = 5,528 (5%)

Sierra Madre
total spruce-fir acres = 85,195
acres with no age data = 38,366 (45%)

Laramie Mountains
total spruce-fir acres = 2,482
acres with no age data = 270 (11%)

Sherman Mountains
total spruce-fir acres = 22,233
acres with no age data = 5,449 (25%)

Figure 19b. Age distribution of spruce - fir forests on the four units of the Medicine Bow National Forest. Bars represent ten year age classes. Source: MBNF RIS

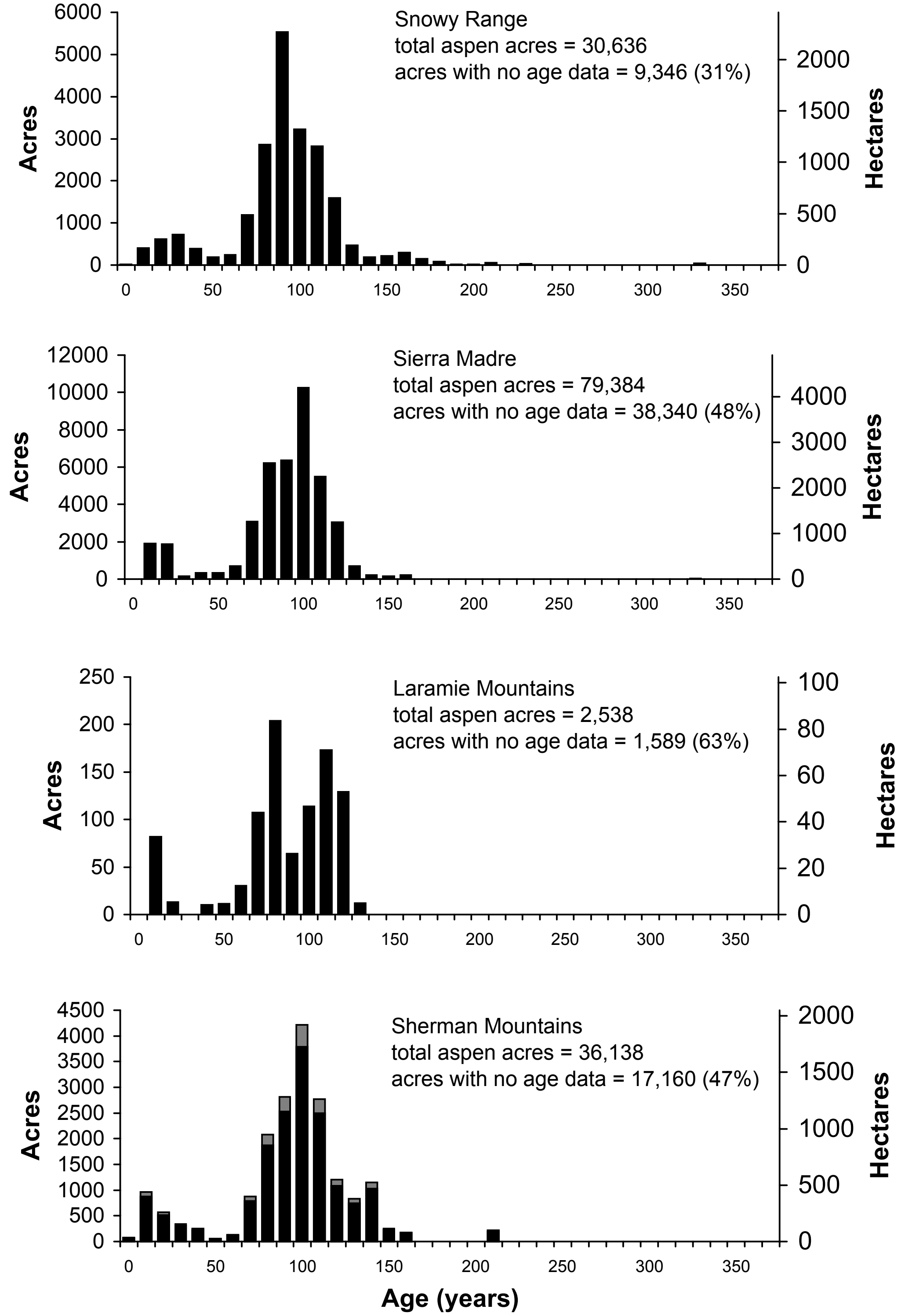


Figure 19c. Age distribution of aspen forests on the four units of the Medicine Bow National Forest. Bars represent ten year age classes. Source: MBNF RI

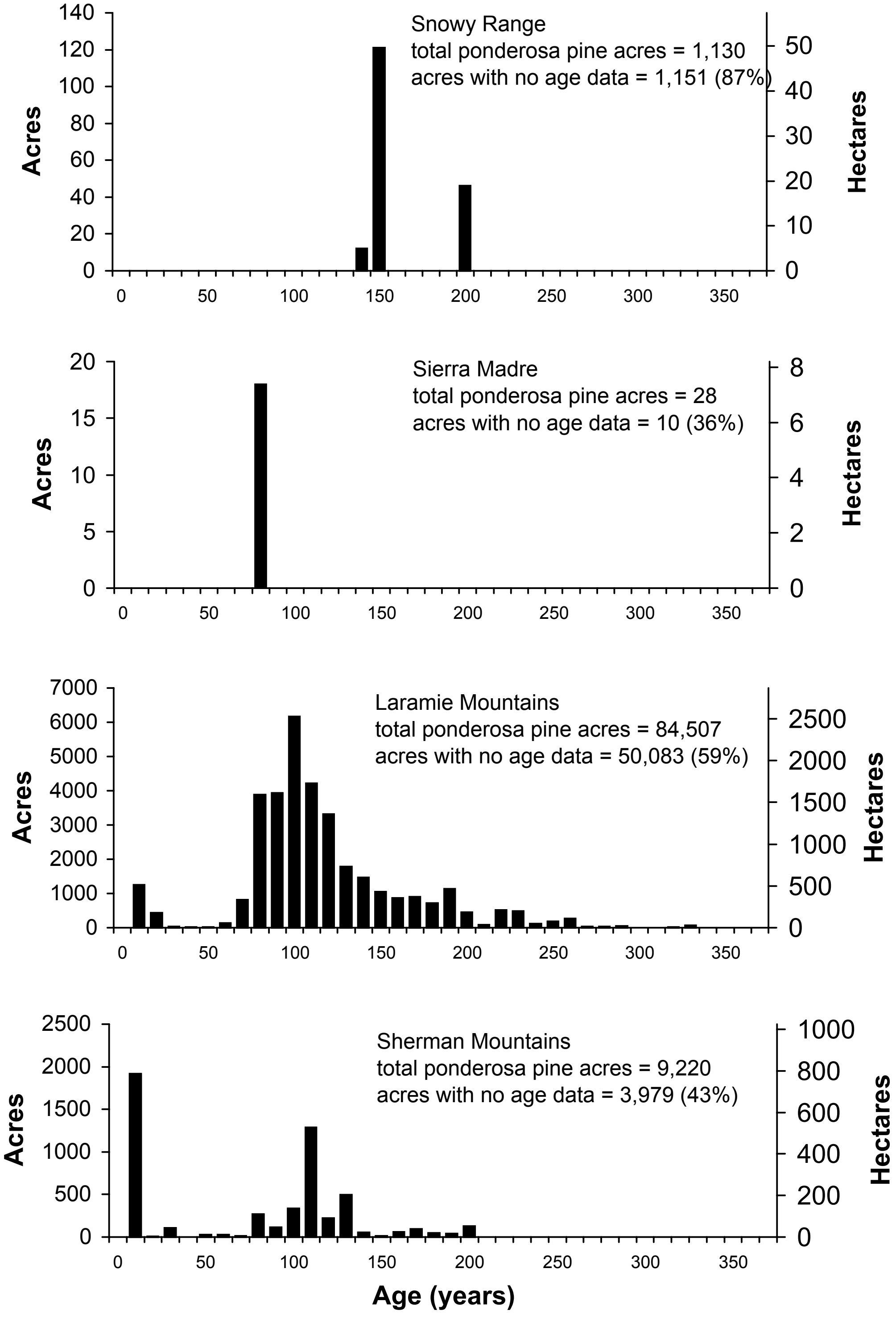
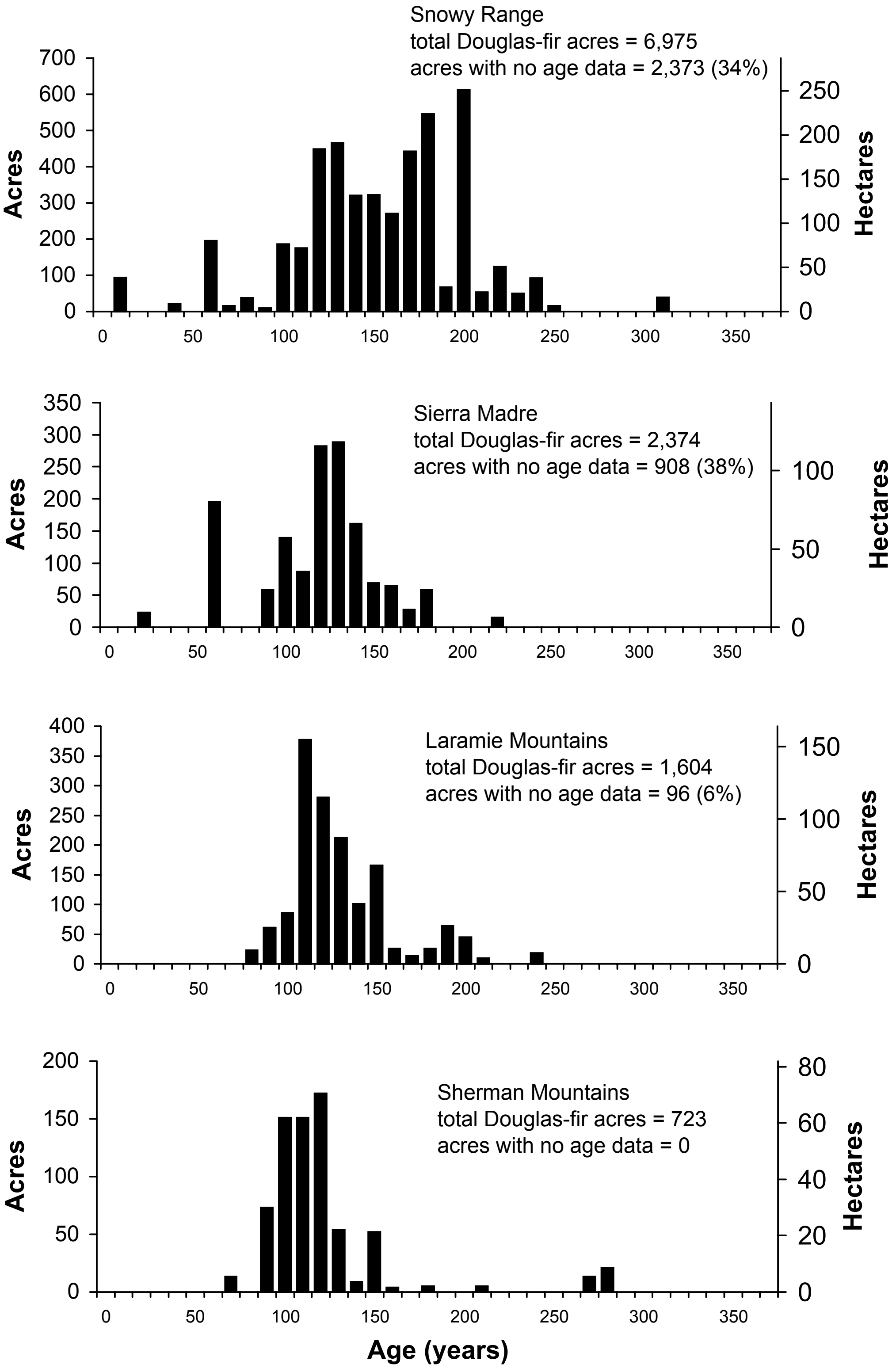


Figure 19d. Age distribution of ponderosa pine forests on the four units of the Medicine Bow National Forest. Bars represent ten year age classes. Source: MBNF RIS


Snowy Range
total Douglas-fir acres = 6,975
acres with no age data = 2,373 (34%)

Sierra Madre
total Douglas-fir acres = 2,374
acres with no age data = 908 (38%)

Laramie Mountains
total Douglas-fir acres = 1,604
acres with no age data = 96 (6%)

Sherman Mountains
total Douglas-fir acres = 723
acres with no age data = 0

Figure 19e. Age distribution of Douglas-fir forests on the four units of the Medicine Bow National Forest. Bars represent ten year age classes. Source: MBNF RIS

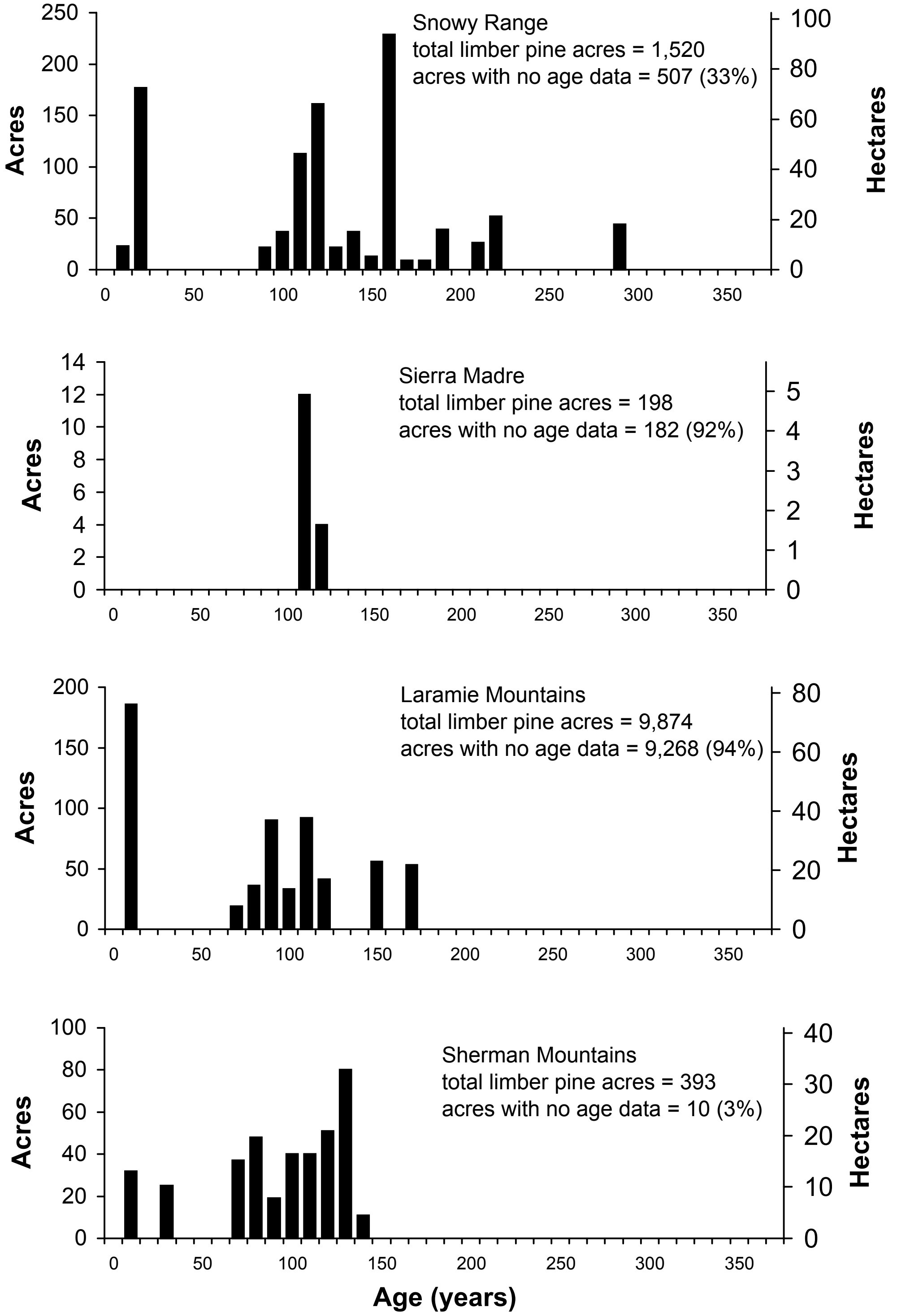


Figure 19f. Age distribution of limber pine forests on the four units of the Medicine Bow National Forest. Bars represent ten year age classes. Source: MBNF RIS

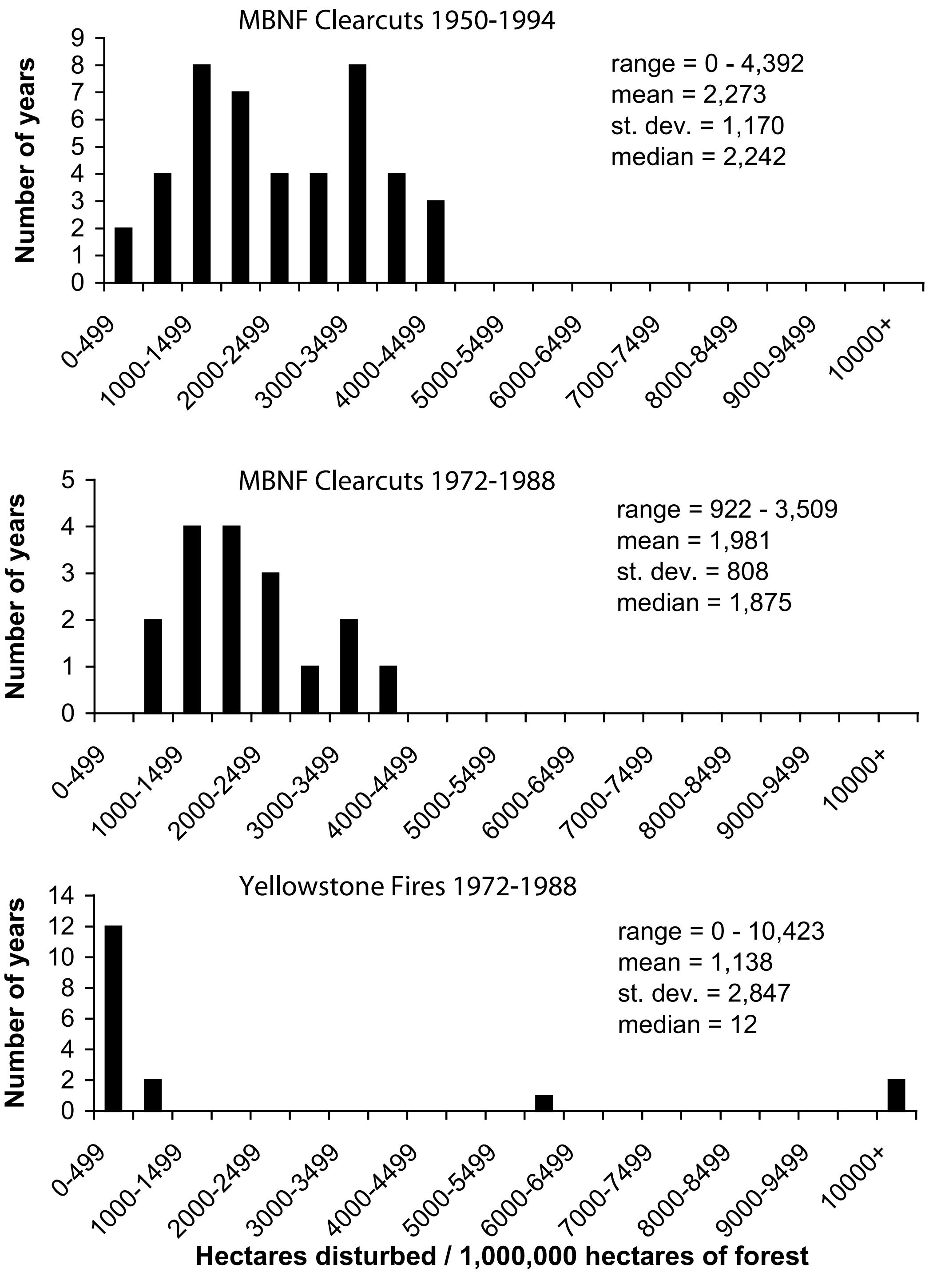


Figure 20. Frequency distributions of area disturbed per year for two different types of stand-replacing disturbance--clearcuts in the MBNF for 1950-1994 and 1972-1988, and fires in Yellowstone National Park from 1972-1988. Data are standardized to hectares per million hectares of forested area. Statistics for YNP were calculated for 1972-1987. If the 321,000 ha burned in 1988 are included, the statistics (standardized) are: range = 0 - 401,250; mean = 24,674; st.dev. = 97,081; median = 16. Sources: von Ahlefeldt and Speas (1996) for MBNF; Despain and Romme (1991) for YNP.

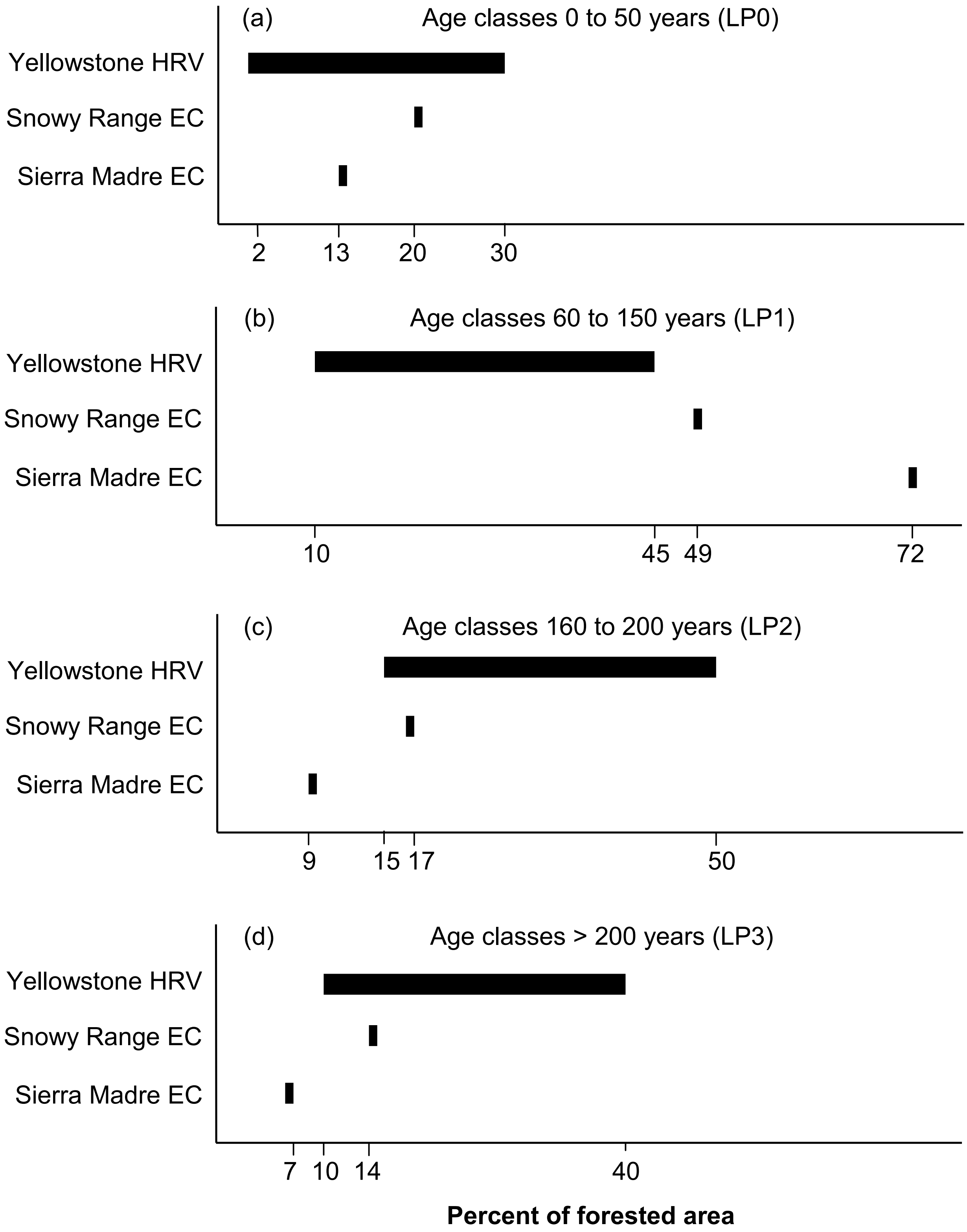


Figure 21. Abundance of early, middle, and late successional stages of subalpine forests on the Snowy Range and Sierra Madre units of the MBNF, compared to historical ranges for the various stages in YNP. MBNF data represent lodgepole pine and spruce/fir forests for which age data are available (147,640 ha of the Snowy Range, including Sheep Mountain, and 66,000 ha of the Sierra Madre; source: MBNF RIS). YNP data represent similar forests within a 129,600 ha study area (Despain and Romme 1991). EC = existing condition.

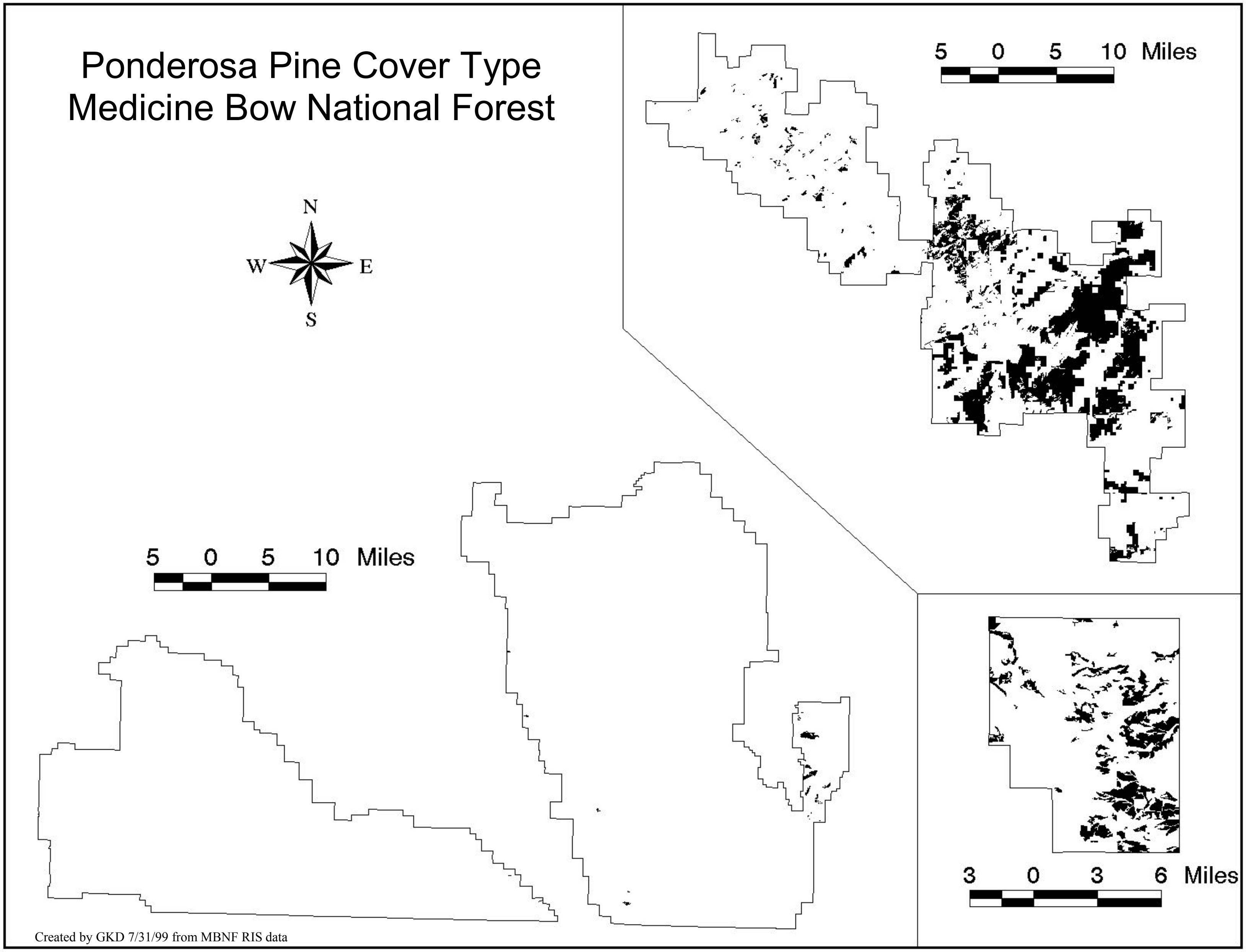

Ponderosa Pine Cover Type
Medicine Bow National Forest
N
W
E
S
5 0 5 10 Miles
5 0 5 10 Miles
3 0 3 6 Miles
Created by GKD 7/31/99 from MBNF RIS data

Figure 22. Distribution of ponderosa pine forests and woodlands on the Medicine Bow National Forest. Source: MBNF

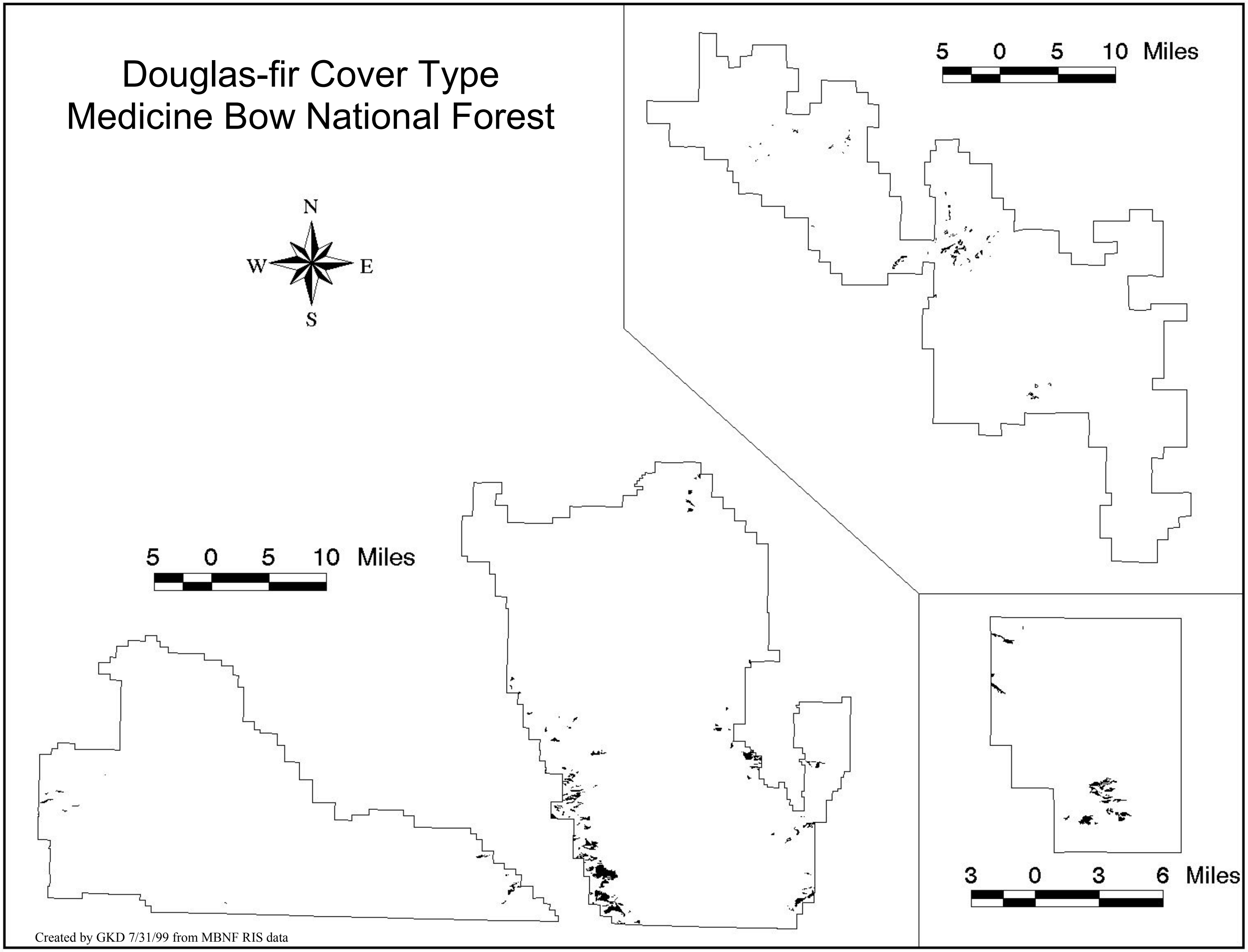

Douglas-fir Cover Type
Medicine Bow National Forest
N
W
E
S
5 0 5 10 Miles
5 0 5 10 Miles
3 0 3 6 Miles
Created by GKD 7/31/99 from MBNF RIS data

Figure 23. Distribution of Douglas-fir woodlands on the Medicine Bow National Forest. Source: MBNF

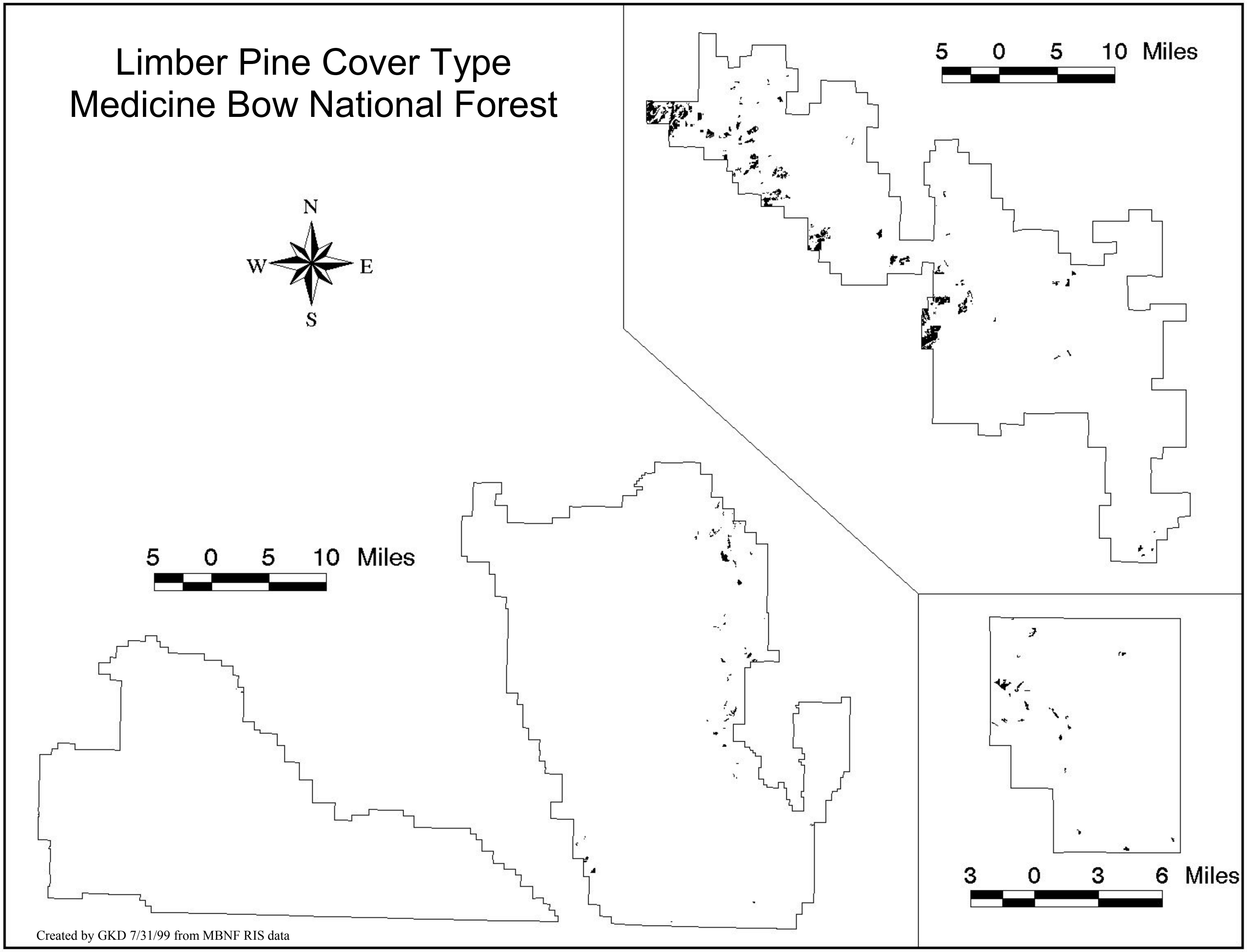

Limber Pine Cover Type
Medicine Bow National Forest
N
W
E
S
5 0 5 10 Miles
5 0 5 10 Miles
3 0 3 6 Miles
Created by GKD 7/31/99 from MBNF RIS data

Figure 24. Distribution of limber pine woodlands on the Medicine Bow National Forest. Source: MBNF

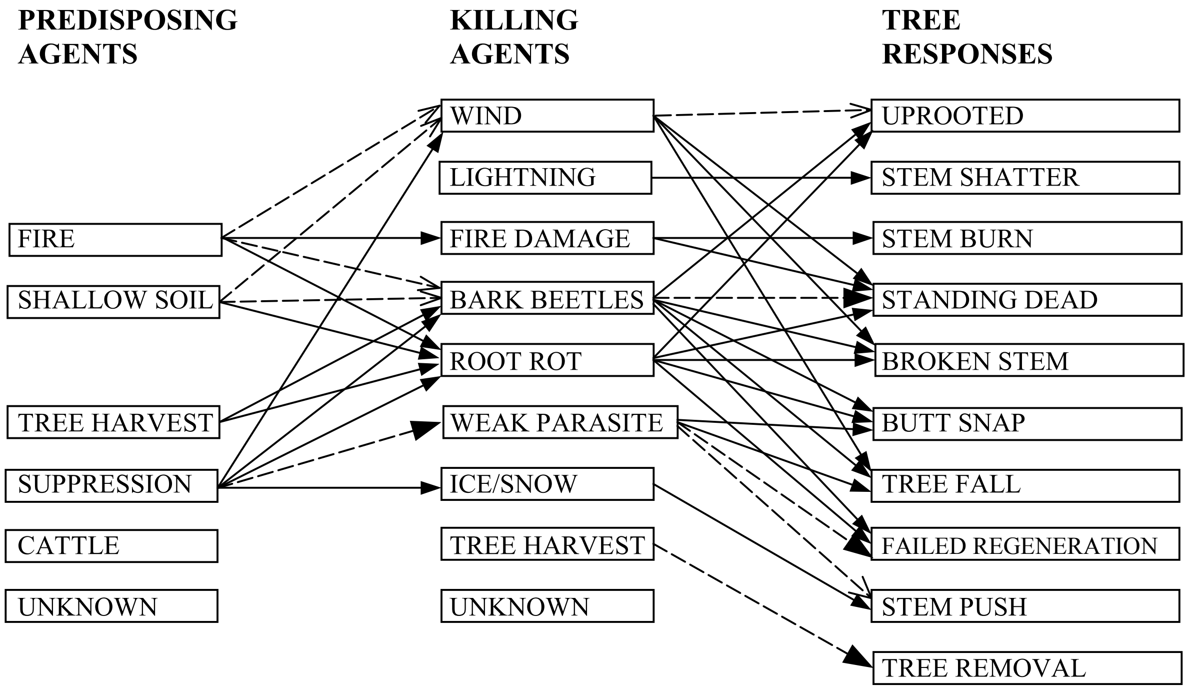

PREDISPOSING AGENTS
KILLING AGENTS
TREE RESPONSES
FIRE
SHALLOW SOIL
TREE HARVEST
SUPPRESSION
CATTLE
UNKNOWN
WIND
LIGHTNING
FIRE DAMAGE
BARK BEETLES
ROOT ROT
WEAK PARASITE
ICE/SNOW
TREE HARVEST
UNKNOWN
UPROOTED
STEM SHATTER
STEM BURN
STANDING DEAD
BROKEN STEM
BUTT SNAP
TREE FALL
FAILED REGENERATION
STEM PUSH
TREE REMOVAL

Fig. 25. Diagram illustrating the interactions among predisposing agents, killing agents, and tree responses observed at three sites in the Black Hills. Dashed arrows with large arrowheads indicate pathways enhanced by timber harvesting. Dashed arrows with open arrowheads indicate pathways diminished by timber harvesting. Solid arrows suggest pathways largely unaffected by timber harvesting. From Lundquist 1995a.

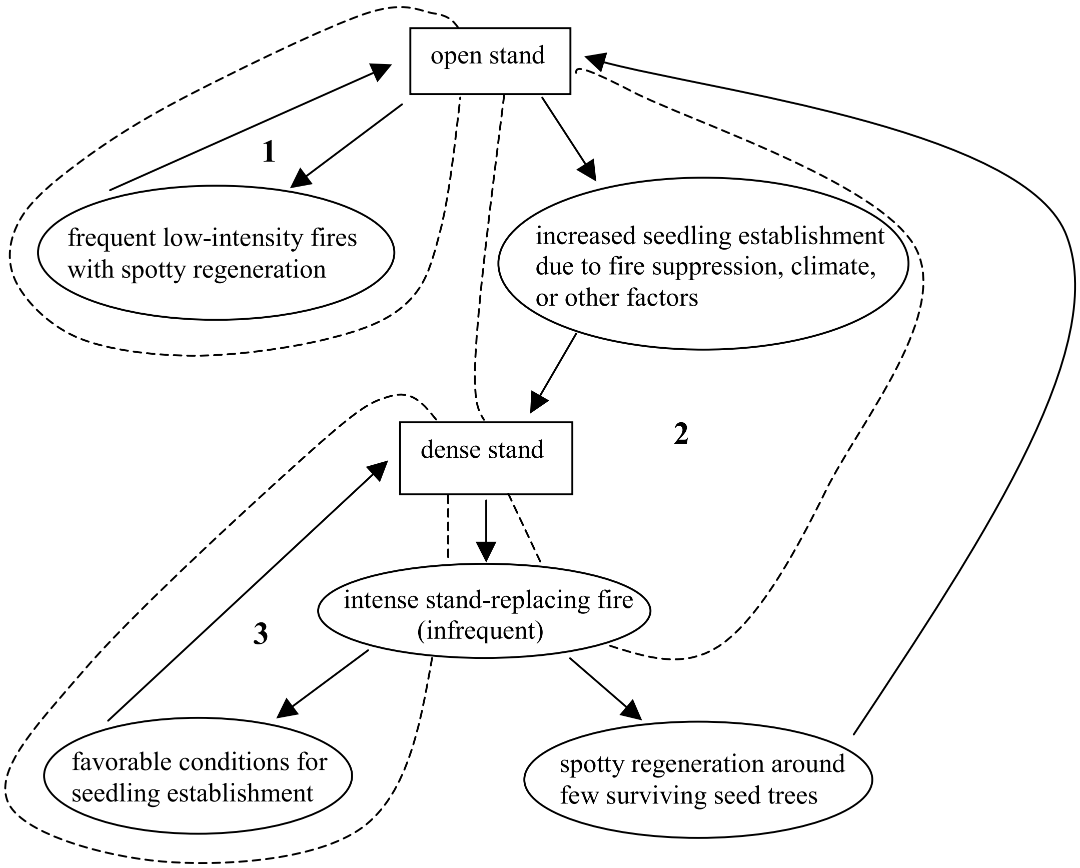

open stand
1
frequent low-intensity fires with spotty regeneration
increased seedling establishment due to fire suppression, climate, or other factors
2
dense stand
intense stand-replacing fire (infrequent)
3
favorable conditions for seedling establishment
spotty regeneration around few surviving seed trees

Fig. 26. Diagram illustrating various pathways of stand development in ponderosa pine ecosystems. Rectangles represent vegetation states, ovals represent processes, and arrows represent changes through time. Interactions among the components enclosed with dashed lines represent the following models or trajectories described in the literature: 1) the equilibrium model predominant in the southwest US and thought to be common on relatively dry sites in Colorado and Wyoming; 2) the trajectory of many ponderosa pine forests during this century; and 3) the nonequilibrium model described for the Black Hills and applicable to more mesic sites in Colorado and Wyoming. See text for discussion and literature citations.

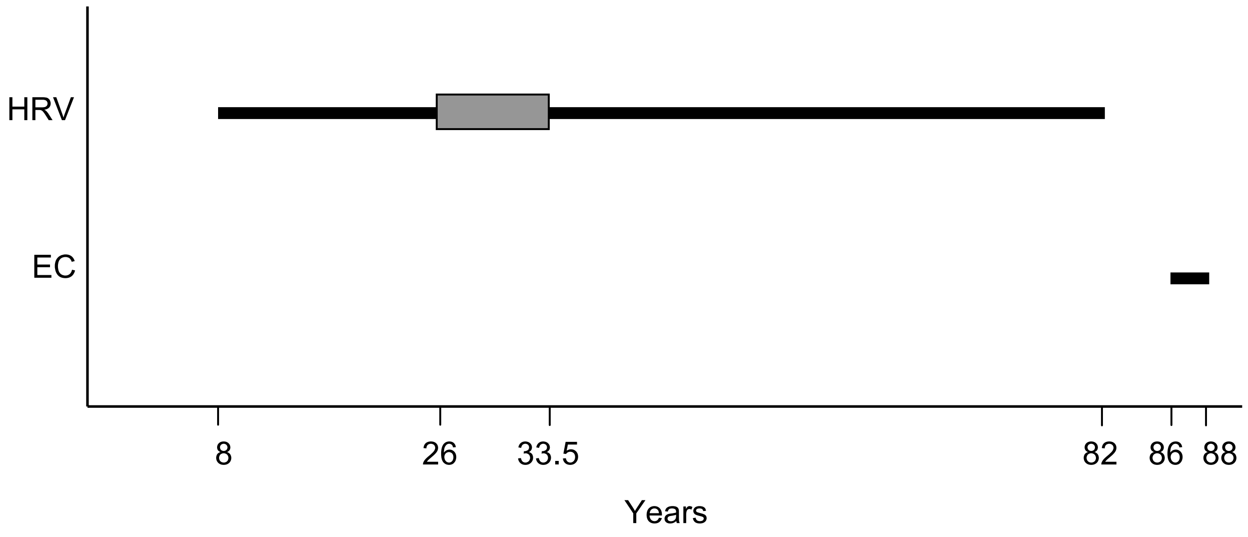


Figure 27. Return intervals for surface fires in stands of ponderosa pine in the Ashenfelder Basin, north of Laramie Peak in the Medicine Bow National Forest. The full bar represents the HRV for fire return intervals prior to ca. 1900, while the shaded area is an estimate of the median interval during the same period. HRV = historic range of variability; EC = existing condition, which is essentially the number of years since the last fire. Data are from Brown et al. (in review).

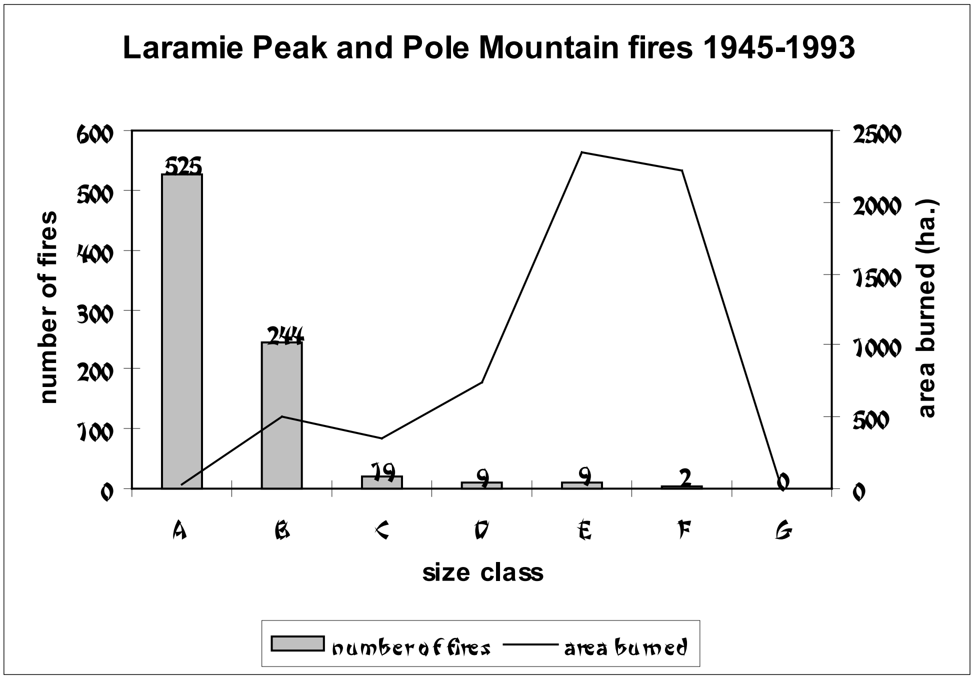


Fig. 28. Number of fires and area burned on the Laramie Mountains and Sherman Mountains units, 1945-1993, on the Medicine Bow National Forest. Based on data presented by von Ahlefeldt and Speas (1996). Size classes are: A, < 0.1 ha.; B, 0.1-3.9 ha.;
C, 4-39 ha.; D, 40-122 ha.; E, 123-409 ha.; F, 410-2049 ha.; G, >2049 ha.

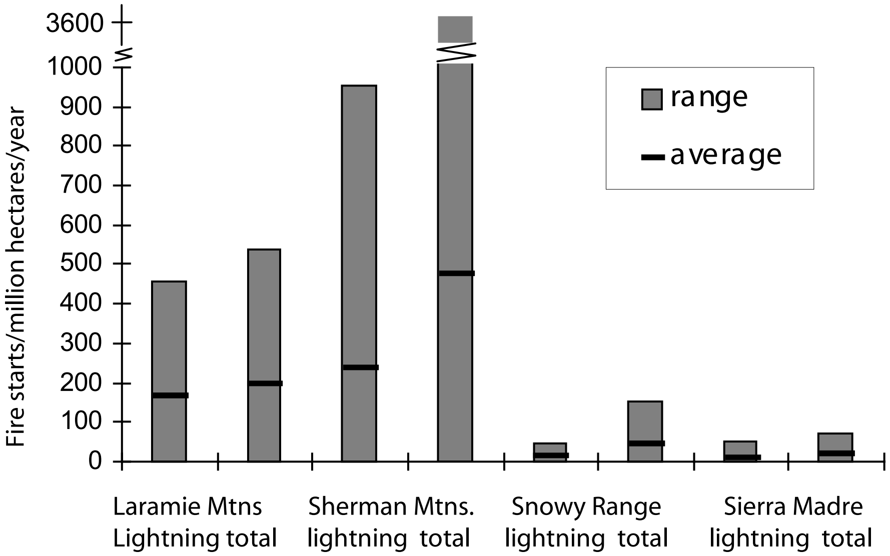


Fig. 29. Comparison of fire starts in the four units of the Medicine Bow National Forest, standardized to fire starts per million hectares per year. The Laramie Peak and Pole Mountain units have mostly low elevation forests (predominantly ponderosa pine). Note that lightning-caused fires are much more frequent in the Laramie and Sherman Mountains than elsewhere. Source: von Ahlefeldt and Speas 1996, based on MBNF fire records for 1945-1993.

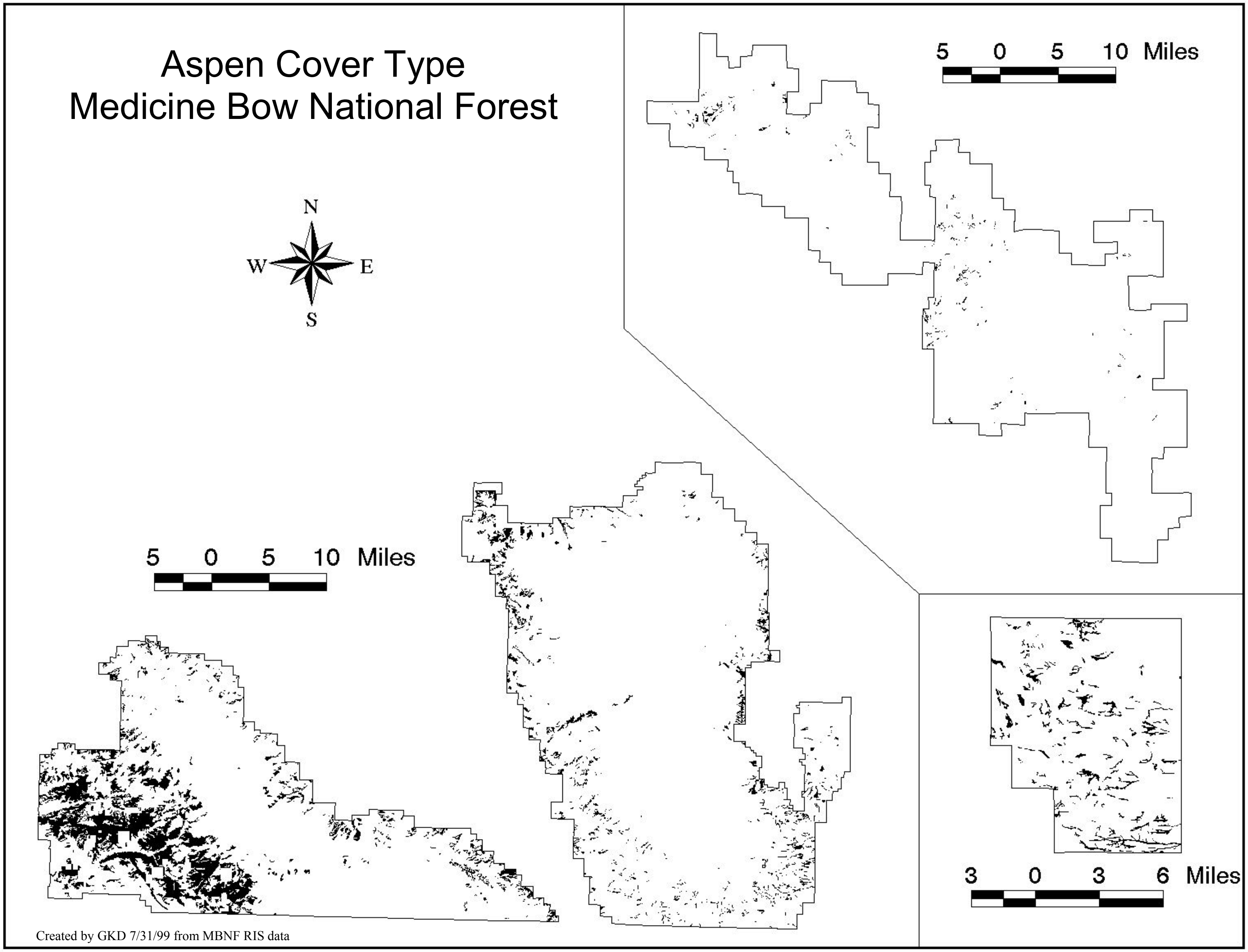

Aspen Cover Type
Medicine Bow National Forest
N
W
E
S
5 0 5 10 Miles
5 0 5 10 Miles
3 0 3 6 Miles
Created by GKD 7/31/99 from MBNF RIS data

Figure 30. Distribution of aspen forests and woodlands on the Medicine Bow National Forest. Source: MBNF